EXPLORING RIPARIAN VEGETATION INTERACTIONS WITH FLOW REGIME AND FLUVIAL PROCESSES FOR AN IMPROVED RIVER MANAGEMENT AND CONSERVATION

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THESIS PRESENTED TO OBTAIN THE DOCTOR DEGREE IN

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Dedico esta tese a meu pai, Vitor Rivaes

Resumo

Examinaram-se os efeitos das alterações do regime de caudais na estrutura e funcionamento da vegetação ripária, bem como, a influência das respetivas alterações da vegetação nas comunidades aquáticas e processos fluviais. Os regimes de caudal expectáveis motivados pelas alterações climáticas originam modificações na vegetação ripária, principalmente a sua redução generalizada em área e maior risco para as fases de sucessão mais jovens e menos tolerantes à tensão hídrica. No contexto europeu, os rios mediterrânicos são os que se encontrarão sob maior ameaça. A perturbação fluvial tem diferentes efeitos na localização e forma dos polígonos de vegetação, sendo o principal condutor da localização dos polígonos, a hidrologia da toalha freática, revelando uma predominante zonagem das fases de sucessão que se sobrepõe à sucessão ecológica natural. Determinou-se o mosaico ripário natural potencial em rios regularizados, permitindo o estabelecimento de condições de referência para caudais ambientais. Os caudais de manutenção ripária são capazes de restaurar a dinâmica do mosaico ripário e reduzir os efeitos da regularização nestas comunidades. A conservação de um caudal mínimo durante o período de estio permite evitar a invasão e amontoamento no canal por parte da vegetação. A alteração do mosaico ripário tem influência nas características hidráulicas do rio, modificando a disponibilidade de habitat das espécies de ictiofauna local de acordo com a magnitude destas modificações. Caudais ambientais que desconsideram os requisitos de vegetação ripária tornam-se obsoletos em poucos anos devido à alteração das premissas de habitat para as quais foram determinados, revelando-se assim insustentáveis a longo prazo e fracassando no alcance dos efeitos desejados nas comunidades aquáticas para as quais foram propostos primeiramente. Considerar os requisitos de vegetação ripária nos caudais ambientais apresenta-se como uma medida primordial para assegurar a eficiência dos regimes de caudais ambientais na perspetiva de longo prazo do ecossistema fluvial.

Palavras-chave

Ecossistemas fluviais, requisitos ripários, caudais de manutenção ripária, ictiofauna, caudais ambientais.

Abstract

This thesis composes an assessment of the flow regime effects on the structure and functioning of riparian vegetation, as well as, the influence of the respective vegetation changes back on the aquatic communities and fluvial processes. The expected flow regimes driven by climate change originate amendments in riparian vegetation, characterized mainly by a general area reduction and a greater menace to the younger and water dependent succession phases. In the European context, the Mediterranean rivers are the ones most threatened. The fluvial disturbance has different effects on the location and shape of the vegetation patches, being groundwater hydrology the main driver of patch location, which exposes a predominant zonation of succession phases over the natural ecologic succession. The potential natural riparian patch mosaic was determined in regulated rivers, allowing for the establishment of reference conditions for environmental flows. The riparian maintenance flows are able to restore the dynamics of riparian vegetation and reduce the effects of regulation in these communities. The safeguarding of a minimum discharge during the summer period allows for the prevention of channel invasion and encroachment by vegetation. The changes in the riparian patch mosaic have influence in the hydraulic characteristics of the river channel, changing the habitat availability of the local fish species according to the magnitude of the changes. Environmental flows that disregard riparian vegetation requirements become obsolete in few years due to the modification of the habitat premises for which they were based, revealing therefore to be unsustainable in the long term and failing to achieve the desired effects on aquatic communities to which those were proposed in the first place. Accounting for the requirements of riparian vegetation into environmental flows poses an essential measure to assure the effectiveness of environmental flow regimes in the long-term perspective of the fluvial ecosystem.

Keywords

Riparian ecosystems, riparian flow requirements, riparian maintenance flows, fish fauna, environmental flows.

Resumo alargado

A tese que se apresenta versa sobre o aprofundamento do conhecimento relativo aos efeitos das alterações do regime hidrológico na vegetação ripária e, consequentemente, no biota aquático e processos fluviais. Não obstante todo o conhecimento científico relativo aos ecossistemas de água doce, existem ainda inúmeras lacunas no conhecimento que motivam a realização desta tese. Nomeadamente, existe ainda pouca disponibilidade de modelos aplicáveis às regiões mediterrânicas usando uma abordagem à escala da paisagem fluvial, bem como uma parca investigação sobre as vias específicas pelas quais estes condutores a afetam. Ademais, no que diz respeito a caudais ambientais, existe ainda uma escassez de incorporação de considerações por espécies tipicamente menos monitorizadas e pelos processos de ligação entre as interfaces aquática e terrestre. A investigação referente às relações entre espécies ripárias e aquáticas é ainda irregular e pontuada pela inexistência de estudos de longo prazo que suportem a compreensão dos efeitos acumulados de longo e curto prazo sobre os ecossistemas fluviais na região mediterrânica. Neste sentido, foram tomados como objetivos da tese: calibrar e validar um modelo dinâmico de vegetação ripária, baseado da relação preditiva entre as guildas de resposta da vegetação ao caudal e o regime hidrológico do rio; avaliar os principais condutores da sucessão ecológica da vegetação ripária, bem como apreciar a sua influência relativa na determinação dos requisitos de caudal deste ecótono; desenvolver uma abordagem inicial de medidas de restauro da vegetação por intermédio da gestão do regime de caudais; definir condições de referência para caudais ambientais; conceber um enquadramento holístico preliminar para caudais ambientais por combinação de requisitos ripários e aquáticos; testar a validação do referido enquadramento em diferentes tipos de regularização; prever as alterações estruturais e funcionais das comunidades ribeirinhas afetadas pelas alterações de caudal a longo prazo; avaliar os efeitos ecológicos de retorno, resultantes da gestão da vegetação, nas comunidades aquáticas.

Na investigação proposta, os modelos apresentam-se como uma ferramenta primordial, coadjuvando os cientistas na melhoria do conhecimento sobre ecossistemas fluviais e munindo os gestores da capacidade de antecipação das implicações ecológicas resultantes de determinada decisão por uma ação de restauro ou forma de gestão. Recorreu-se ao modelo dinâmico de vegetação ripária *CASiMiR-*

vegetation, que permite simular a dinâmica sucessional desta vegetação com base na relação existente entre os elementos do regime hidrológico ecologicamente relevantes e métricas de vegetação refletindo claramente as suas respostas à alteração hidrológica. Utilizou-se também o modelo *River2D*, um modelo bidimensional de elementos finitos incorporando um modelo hidrodinâmico e outro de habitat, para simular as condições de escoamento num determinado troço de rio e estimar o seu valor de habitat segundo a preferência das espécies aquáticas consideradas. O modelo *River2D* foi empregue para o cálculo de diversos parâmetros hidráulicos e disponibilidades de habitat nos troços de rio estudados, apoiando as modelações ecológicas e análises realizadas.

Para alcançar os objetivos propostos na tese examinaram-se os efeitos das alterações do regime de caudais, com diversas origens, na estrutura e funcionamento da vegetação ripária, caracterizada pela sua dinâmica sucessional e atributos da paisagem. Posteriormente, analisou-se a influência das respetivas alterações da vegetação nas comunidades aquáticas, no transporte de sedimento entre outras questões relacionadas. Neste sentido, a tese encontra-se constituída por oito capítulos divididos em seis secções. Na secção I providencia-se a informação introdutória de fundo necessária à compreensão do tema, dos objetivos da tese e da investigação levada a cabo. As seguintes quatro secções são destinadas à apresentação da investigação concretizada para suportar a discussão e conclusões da tese, apresentadas na secção VI.

Designadamente, na secção II estuda-se o efeito dos fatores de larga escala na vegetação ripária, tais como a sazonalidade e a variabilidade da precipitação, considerados como os principais condutores do regime hidrológico e, consequentemente, da sucessão ecológica ripária. Esta influência é avaliada através de uma análise comparativa entre a paisagem ripária resultante do regime hidrológico alterado pelos padrões de precipitação esperados em cenários de alteração climática, e o mosaico ripário natural atual. Aqui, o capítulo 2 apresenta a primeira tentativa conhecida de teste da performance do modelo *CASiMiR-vegetation* num sistema semiárido, bem como da sua capacidade em prever a resposta espaciotemporal da vegetação ripária em função de diferentes regimes de caudais esperados em cenários de alteração climática. No capítulo 3 revela-se uma avaliação similar mas inserida num contexto Europeu, comparando-se as diferentes condições climáticas e

hidrogeomórficas, de modo a providenciar uma visão mais ampla das possíveis respostas deste ecótono face a alterações climáticas. Esta análise faculta ainda uma melhor compreensão da circunstância específica a que está sujeita a vegetação ripária localizada em ecossistemas Mediterrânicos.

Na secção III analisam-se os principais condutores do mosaico de vegetação originados pelo regime de caudais, considerando-se nesta influência dois fatores fundamentais, nomeadamente, a hidrologia da toalha freática e a perturbação morfodinâmica. Particularmente, o capítulo 4 investiga o efeito destas perturbações em dois elementos centrais da ecologia da paisagem, a localização e a forma dos polígonos de vegetação. O efeito dos componentes do regime de caudais nos atributos da paisagem ripária é ilustrado pela quantificação e detalhe dessas influências ao nível da fase de sucessão.

A secção IV engloba dois estudos que exploram potenciais opções de gestão da vegetação em rios regularizados, com vista ao seu restauro e conservação. Para além disso, tenta-se estabelecer uma metodologia para a determinação dos requisitos da vegetação ripária que garantam a sua qualidade ecológica de longo prazo em rios regularizados. Assim, no capítulo 5 apresentam-se várias simulações da resposta da vegetação a diferentes regimes de caudais de manutenção ripária, com o intuito de determinar o melhor regime para o restabelecimento da dinâmica do mosaico ripário e redução dos efeitos da regularização de caudais na vegetação ripária a jusante de barragens. No capítulo 6 divulga-se a primeira tentativa de aplicar e validar técnicas de modelação de habitat hidráulico à vegetação aquática, de forma a determinar um caudal anual mínimo com a capacidade de reduzir o risco de invasão e amontoamento no canal por espécies exóticas num rio mediterrânico altamente regularizado.

A secção V é dedicada à apreciação dos efeitos da gestão da vegetação nas comunidades aquáticas. Mais precisamente, no capítulo 7 avaliam-se as consequências de considerar, ou não, os requisitos de vegetação ripária em caudais ambientais, no que diz respeito à qualidade do habitat de espécies piscícolas. Consequentemente, avalia-se a eficiência dos regimes de caudais ambientais na perspetiva de longo prazo do ecossistema fluvial. Esta tarefa foi abordada do ponto de vista ecohidráulico, de uma forma inovadora com grande potencial para a predição dos ajustamentos da vegetação aquática em ecossistemas fluviais.

Por fim, a secção VI apresenta uma discussão geral dos resultados da tese e consequentes conclusões. Neste tomo, verifica-se que os objetivos da tese foram amplamente alcançados através da investigação realizada e apresentada nos capítulos anteriores. Especificamente, em relação aos condutores de larga escala da vegetação ripária, foi possível determinar que os expectáveis regimes de caudal motivados pelas alterações climáticas originam modificações na vegetação ripária, com especial enfoque para a sua generalizada redução em área. As fases de sucessão mais jovens e dependentes da água serão a mais afetadas pelas alterações climáticas, com modificações dramáticas que poderão implicar uma séria ameaça à viabilidade de espécies em particular. No referido contexto europeu, destaca-se ainda a situação de maior perigo a que estão votados os rios mediterrânicos.

Relativamente aos fatores de pequena escala da vegetação ripária, ficou demonstrado que a perturbação fluvial tem diferentes efeitos na localização e forma dos polígonos de vegetação. O principal condutor da localização dos polígonos é a hidrologia da toalha freática, revelando uma predominante zonagem das fases de sucessão que se sobrepõe à sucessão ecológica natural. Por outro lado, a forma dos polígonos não aparenta ser especialmente determinada pela perturbação fluvial embora, na sua pouca influência, tanto a hidrologia da toalha freática como a perturbação morfodinâmica apresentem contributos substanciais. Estes resultados realçam a necessidade de procedimentos próprios durante a gestão do regime de caudais, mais concretamente, a manutenção de um caudal mínimo de recarga dos níveis freáticos.

No âmbito da gestão da vegetação ripária foi ainda possível determinar o mosaico ripário natural potencial em rios regularizados, permitindo o estabelecimento de condições de referência para caudais ambientais. Os caudais de manutenção ripária mais adequados são capazes de restaurar a dinâmica do mosaico ripário e reduzir os efeitos da regularização nas comunidades ripárias a jusante de barragens. Este resultado habilita o desenvolvimento de uma abordagem inicial ao restauro da vegetação ripária por meio da gestão do regime de caudais. Além do mais, a manutenção de um caudal mínimo no rio durante o período de estio permite evitar a invasão e amontoamento do canal por parte de espécies aquáticas invasoras. Mesmo que este não leve à sua total eliminação, reforça seguramente a performance competitiva das espécies autóctones.

No que diz respeito à análise dos efeitos de retorno da gestão da vegetação, verifica-se que a alteração do mosaico ripário influencia as características hidráulicas do rio, modificando a disponibilidade de habitat das espécies de ictiofauna local de acordo com a magnitude destas modificações. Constatou-se desta forma que caudais ambientais desconsiderando os requisitos de vegetação ripária se tornam obsoletos em poucos anos devido à alteração das premissas de habitat para as quais foram determinados, tornando-se assim insustentáveis a longo prazo e fracassando no alcance dos efeitos desejados nas comunidades aquáticas para as quais foram propostos primeiramente. Assim sendo, considerar os requisitos de vegetação ripária nos caudais ambientais apresenta-se como uma medida primordial para assegurar a eficiência destes na perspetiva de longo prazo do ecossistema fluvial.

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ANOVA	Analysis of Variance
AUC	Area Under the Curve
BBM	Building Block Methodology
BF	Bog Forest phase
CFI	Comparative fit index
CIS	Common implementation strategy
CO2	Carbon dioxide
CSI	Composite Suitability Index
CSI	Composite Suitability Index
DBH	Diameter at Breast Height
DEM	Digital Elevation Model
DIPSIR	Driver-Pressure-State-Impact-Response framework
DO	Deep Oxbow phase
DRIFT	Downstream Response to Imposed Flow Transformation
DSI	Depth Suitability Index
DSI	Depth Suitability Index
EF	Established Forest phase
Eflow&Flush	Environmental flow addressing fish and riparian requirements
Eflow	Environmental flow
EFP	Established forest phase
ELOHA	Ecological Limits of Hydrologic Alteration
ES	Early Successional Woodland phase
ESWP	Early Successional Woodland phase
EU	European Union
GCM	Global Circulation Model
GFI	Goodness of fit index
GHG	Greenhouse gas
GPS	Global Positioning System
grndh	Groundwater hydrology
GWDi	Groundwater depth index
HHS	Hydraulic Habitat Suitability
HP*	Herb Reed Phase
HP	Herb Woodland

HSC	Habitat Suitability Curve
IDF	Intensity-Duration-Frequency
IFIM	Instream Flow Incremental Methodology
IP	Initial Phase
ku	Kurtosis
MAD	Mean Absolute Deviation
MADP	Mean Absolute Percentage Deviation
MDi	Morphodynamic disturbance index
Med-region	Mediterranean region
Med-rivers	Mediterranean rivers
MF	Mature forest phase
MFP	Mature forest phase
MPFD	Mean patch fractal dimension
mrphd	Morphodynamic disturbance
PCA	Principal Component Analysis
PERIMTR	Perimeter
Phslc	Patches location
Phssh	Patches shape
PP	Pioneer phase
RCHARC	Riverine Community Habitat Assessment and Restoration Concept
RCM	Regional Circulation Model
RF	Random forest algorithm
RMSD	Root Mean Square Deviation
RMSE	Root Mean Square Error
RMSEA	Root mean square error of approximation
SCS	Soil Conservation Service
SEM	Structural equation modeling
sk	Skewness
SNIRH	Portuguese Water Resources National Information System
SO	Shallow Oxbow phase
SP*	Shrub Reed Phase
SP	Shrub Woodland Phase
SRES	Special Report on Emissions Scenarios
SRMR	Standardized root mean square residual

SSI   Substrate Suitability Index

THAD   Distance to thalweg

THAH   Height to thalweg

TL   Total Length

TLI   Tucker-Lewis Index

TSS   True Skill Statistic

UF   Upland Forest

VIF   Variance inflation factor

VSI   Velocity Suitability Index

VSI   Velocity Suitability Index

WFD   Water Framework Directive

WMO   World Meteorological Organization

WUA   Weighted Usable Area

YSWP   Early Successional Woodland phase

SECTION I

INTRODUCTION

CHAPTER 1

Introduction

1. Background

1.1. The hydrosphere

The Earth's diversity is a result of four main spheres – atmosphere (air), lithosphere (land), hydrosphere (water) and biosphere (living organisms) – that interconnectedly compose the surface of the earth (Rosenberg, 2017). The hydrosphere is the Earth's discontinuous envelope of water that lies between the atmosphere and the lithosphere. Its volume reaches approximately 1.4 billion cubic kilometers, comprising all the water in the oceans, the inland surface water and groundwater (L'vovich, 1979). Water can be found everywhere on Earth and is an essential compound for the existence of life as we know it (Ball, 2001). It is no coincidence that when scientists look for life in other planets they search primarily for the existence of water (Rasmussen, 2017). Notwithstanding, the available freshwater (not in the form of glaciers, permanent snow and permafrost) is only a very small fraction, approximately 0.8% of the total water reserves (Table 1).

Table 1. The volume of the hydrosphere (adapted from: Shiklomanov, 1993).

Components of the Hydrosphere	Volume of water (10^3 km^3)	Percent of total volume
Salt water		
World ocean	1 338 000	96.54
Saline groundwater	12 853.5	0.93
Salt water lakes	85.4	0.006
Total salt water	1 350 938.9	97.47
Freshwater		
Glaciers, permanent snow cover and permafrost	24 364	1.76
Groundwater	10 546.5	0.76
Lakes and swamps	102.47	0.007
River water	2.12	0.0002
Biological water	1.12	0.0001
Atmospheric water	12.9	0.0009
Total freshwater	35 029.11	2.53
Total water reserves	1 385 968	100

The hydrosphere is a very dynamic element. The water contained in it is in constant movement, powered by its driving force, the hydrologic cycle. The uneven Earth's heating generates convective movements of air that are responsible for the water evaporation and condensation in the atmosphere. Later, the force of gravity takes care of the phenomena of precipitation, infiltration, runoff and convection currents (Musy and Higy, 2011). Water is therefore primarily evaporated and desalinated from the oceans and delivered as freshwater in inland in the form of precipitation. Freshwater then returns to the oceans mainly by surface runoff or groundwater flow, originated by the drainage of the watersheds (Figure 1). The watershed is the main spatial unit in hydrology and stands for an area in which all the water arriving it congregates to the same convergence point – the outlet (Musy and Higy, 2011). Consequently, all the water flowing superficially will contribute for the surface water in rivers, lakes and ponds that flow into the oceans (in the case of exorheic watershed) or to closed hydrologic systems (in the case of endorheic watershed) originating permanent or temporary inland surface water bodies.
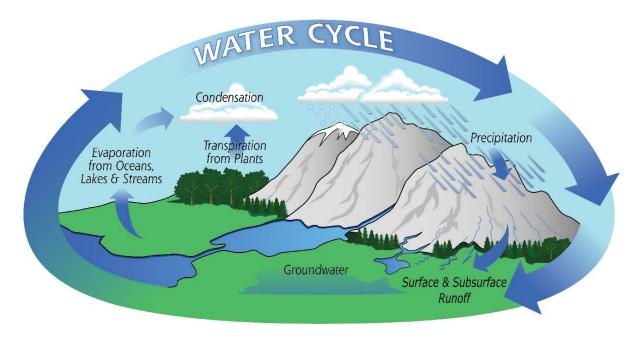
Figure 1. The hydrologic cycle (adapted from: https://pmm.nasa.gov/education/water-cycle).

1.2. The freshwater system

Freshwater is a very small part of the hydrosphere in which water presents low concentrations of dissolved salts. It can be found in the liquid state in lakes and reservoirs, rivers, groundwater, soil moisture and vapors in the atmosphere. Freshwater ecosystems cover less than one hundredth of the Earth's surface but are estimated to shelter more than 100 000 species, i.e., approximately 6% of all the described species on Earth (Dudgeon et al., 2006).

Despite of the derisive volume of freshwater comparing to the total amount of existing water, it is the most important limiting factor for plant growth and for ecosystem production on land (Moss, 2010), as well as extremely important in the provision of vital services for human existence (Aylward et al., 2005). As a provision service, freshwater delivers not only water for human consumption but also for non-consumption usage along with food and medicines (Aylward et al., 2005). It also delivers regulatory services such as the maintenance of water quality, floods and droughts mitigation, as well as the conservation of biodiversity. Freshwater ecosystems are also responsible for cultural services like recreation, tourism, and existence values like provision of beauty and life fulfilling values. In addition, human existence also benefits from nutrient cycling and ecosystem resilience provided by the freshwater supporting services (Postel and Richter, 2003; Aylward et al., 2005). All together, these circumstances demonstrate that the conservation of such important reserve is paramount for human existence.

1.3. Threats to freshwater systems

Despite all of the mentioned previously, freshwater ecosystems are on top of the world's most threatened ecosystems (Revenga et al., 2000; Dudgeon et al., 2006), primarily due to river damming (Allan and Castillo, 2007). The threat of biodiversity lost is intensified to downstream as anthropogenic and natural disturbances increase (Bejarano et al., 2011; Bruno et al., 2014). Particularly in the Mediterranean region (Med-region), freshwater ecosystems face current pressures that pose an increased menace due to additional degradation by the amplified exposition to several factors like water-allocation conflicts (Thomas, 2008), drought (Lake, 2003), increased flow regulation (Bejarano et al., 2012), water extraction (Horner et al., 2009), salinization

(Hancock et al., 1996; Pinder et al., 2005), bushfires (Verkaik et al., 2013), invasive species (Hermoso and Clavero, 2011; Aguiar and Ferreira, 2013), human activity (Aguiar et al., 2006; Schnitzler et al., 2007) and habitat fragmentation (García-Ruiz et al., 2011; von Schiller et al., 2011).

In the last decade of the 20th century, the urban area increased 13% across the Mediterranean biome, being particularly felt in the Mediterranean Basin, which underwent the major change, approximately 17% (Underwood et al., 2009). In the future, all these pressures will be intensified and complemented with the desertification resulting from the expansion of semi-arid and arid systems (Parry et al., 2007; Klausmeyer and Shaw, 2009). By 2100, the Mediterranean biome is expected to experience the largest proportional loss of biodiversity of all inland biomes due to its significant sensitivity to multiple biodiversity threats and interactions among them (Sala et al., 2000). For instance, above 60% of the vegetation will experience a reduction of more than 80% within this century (Klausmeyer and Shaw, 2009).

Accordingly, the livelihood and survival in the freshwater systems will be greatly affected since pollution, water extraction, invasive species, reservoirs, agriculture and overfishing are considered to be the main threats to fish species (Hermoso and Clavero, 2011). Pollution is especially important for fish fauna due to the strong natural and human stressors found in Mediterranean rivers (Med-rivers) (López-Doval et al., 2013). Likewise, the existing remnants of riparian woodlands (Blondel et al., 2010) are currently under the threat of three main pressures, namely, river damming, invasive species and climate change (Poff et al., 2011).

1.4.   Rivers

The freshwater superficial flows are mostly performed through rivers, which are lotic systems with dynamics and hierarchies working in a four-dimensional perspective (Ward, 1989). Those four dimensions describe the longitudinal, lateral, vertical and temporal phenomena of the river systems (Figure 2). The longitudinal dimension (upstream-downstream continuity) of rivers refers to the biotic and abiotic patterns alteration along the river length. The lateral dimension (river-floodplain interactions) represents the relations between the stream channel, parafluvial zones (gravel and sand bars sideways to the surface stream which are still part of the active channel) and

the adjacent floodplain systems, viewed as critical for maintaining river productivity (Junk et al., 1989) and biotic diversity (Amoros and Bornette, 2002). The vertical dimension (river-hyporheic zone connection) is a much less studied perspective that highlights the interaction between surface and groundwater flows, particularly in the hyporheic zone (the volume of river substrate immediately under the riverbed surface where shallow groundwater and surface flow interrelate). Finally, the temporal dimension (moment and timing of events) is translated as the different time scales in ecological systems that determine the temporary and longstanding changes and dynamics of the lotic systems.
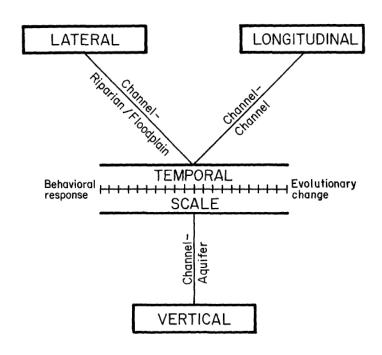
Figure 2. The four-dimensional perspective of rivers (from: Ward, 1989).

Together, these four dimensions of the lotic system influence the ecosystem processes and patterns in rivers at multiple scales, inducing river biotic productivity and diversity (Stanford and Ward, 1988; Junk et al., 1989; Amoros and Bornette, 2002), impelling fish communities (Belliard et al., 1997; Torgersen et al., 2006; Araújo et al., 2008; Burgess et al., 2013; Kawanishi et al., 2013), aquatic invertebrates (Malard et al., 2003; Winkelmann et al., 2003; Freitag, 2004; Arscott et al., 2005; Paillex et al., 2007; Paillex et al., 2009; Vieira and Fonseca, 2013; Holt et al., 2015), water physical chemistry (Lewis et al., 2006; Lewis et al., 2007; von Schiller et al., 2008; von Schiller et al., 2011; Batlle-Aguilar et al., 2014; Hug Peter et al., 2017), sediment transport (Powell, 1998;

Sivakumar and Chen, 2007; Singer, 2008; Gurnell et al., 2012; Corenblit et al., 2014; Gurnell et al., 2016) or riparian vegetation (Lite et al., 2005; Naiman et al., 2005; Friedman et al., 2006; Tánago and Jalón, 2006).

The different spatiotemporal scales associated with watershed geomorphic features and events organize hierarchically the natural variability of river systems and their habitats (Frissell et al., 1986). By these means, the different dimensions of the riverine habitat (e.g., segment, reach, pool/riffle and microhabitat) are expected to occur for particular spatial and temporal contexts that determine the systems behavior and capabilities (Figure 3).

STREAM SYSTEM	SEGMENT SYSTEM	REACH SYSTEM	"POOL/RIFFLE" SYSTEM	MICROHABITAT SYSTEM
10^3 m	10^2 m	10^1 m	10^0 m	10^{-1} m
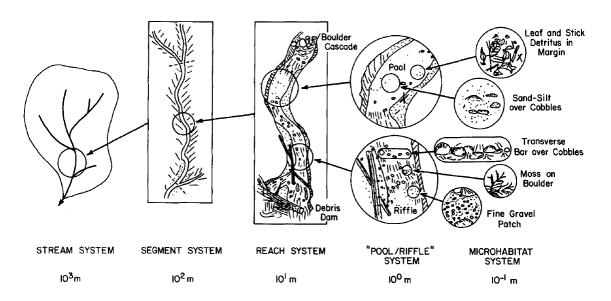
Figure 3. Hierarchical organization of a river system and approximate linear space scales (from: Frissell et al., 1986).

Nevertheless, such hierarchy works in a continuum as the entire system is functionally connected and in permanent transformation as it flows downstream (Vannote et al., 1980). Therefore, depending on the distance to the river source, the closely linked watersheds, floodplains and river systems contribute differently to shape the biological communities, which change in consonance (Figure 4). Along this river continuum, there is also a downstream transport of nutrients by river flow, known as the nutrient spiraling (Newbold et al., 1982). Herein, the organic matter decomposition occurs along the way, taking more or less time according to the spiral length. The invertebrate consumers play here a very important role in controlling spiraling.
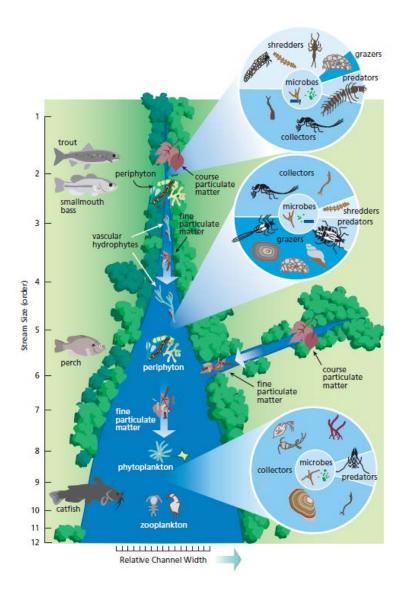
Figure 4. The river continuum concept, as theorized by Vannote et al., 1980 (from: FISRWG, 1998).

All together, these concepts determine a hierarchical patchy arrangement of the riverine landscapes, characterized by different extents of eco-hydrogeomorphic processes in what can be described as the Riverine Ecosystem Synthesis (Thorp et al., 2006). As a result, along the river system, several functional process zones are generated longitudinally as a function of the magnitude of each of the combined ecologic and hydrogeomorphic processes (Figure 5).
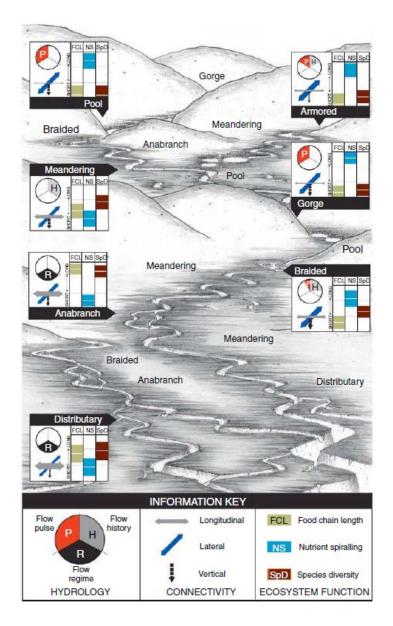
Figure 5. The River Ecosystem Synthesis (Thorp et al., 2006), in which multiple functional process zones, resulting from the combination of eco-hydrogeomorphic processes, occur with different magnitudes in the river system (from: Thorp et al., 2008).

1.5. The river flow regime

The flow regime is the hydrological signature of a river and stands for the variability of river flows found in any river section. The flow unsteadiness results from the hydrological variability pattern created by the interaction between the climatic regime (precipitation and temperature) in the watershed and the watershed features that regulate runoff (Belmar et al., 2010).

The natural flow regime, characterized by its components of magnitude, frequency, duration, timing and rate of change, is primordial for sustaining the ecological integrity

of rivers (Poff et al., 1997). Such natural flow regime paradigm for river conservation and restoration introduces the concept of intra and interanual flow variability as the main driver of native biodiversity and ecological integrity of the river system. Rivers have a natural flow regime, on the basis of which aquatic and riparian communities have evolved and rely the ecological integrity of their ecosystems (Poff et al., 1997). Accordingly, the natural variability of the flow regime is a prerequisite for the maintenance of instream and floodplain dynamics and quality. The river stage is responsible for providing several geomorphic and ecological functions according to the different stages of discharge. The infiltration maintains the water tables that, together with the floods, sustain riparian vegetation. On the other hand, the different floods varying in magnitude and timing maintain the diversity of aquatic and riparian species, as well as the habitats. Accordingly, the occurrence of small floods is very frequent and responsible for the prevention of fine material deposition in the channel bed and maintaining the spawning habitat for fish. Floods with higher recurrence interval are responsible for the sediment deposition in the floodplains, providing newer grounds for the establishment of riparian vegetation and active channel maintenance as well. Less frequent floods, such as the ones with ten years return period, are able to perform the previous tasks and even influence the older successional stages of vegetation in the outer part of the river floodplains. Finally, the very large floods are able to modify completely the river landscape providing large woody debris to the aquatic system, improving habitat and maintaining the river dynamics (Figure 6).
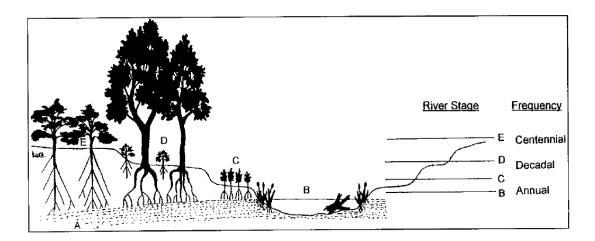
Figure 6. The hierarchy of ecological functions provided by the variability of the flow regime (from: Poff et al., 1997).

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1.6. The river landscape

Landscape ecology is the science that studies the relationships between spatial patterns and ecological processes (Turner and Gardner, 2015). In this context, river corridors are major structural components, following a dendritic pattern progressively widening to downstream (Forman and Godron, 1981) and forming a particular landscape inside the landscape – river landscapes or riverscapes. These are very dynamic landscapes, responding to the flow regime (Miller et al., 1995), particularly to flood pulse in extension, composition and configuration (Ward et al., 2002). Within the riverscapes, the riparian landscapes, seen as the longitudinal and transverse spatial configurations of riparian zones, respond also to the flow regime, integrated in cascades of water, sediment, nutrients and carbon (Malanson, 1993).

There are four major drivers of riverscape pattern: the abiotic template (e.g., climate, landform, soils), the biotic interactions (e.g., competition, influence of dominant organisms, trophic cascades), human land use (e.g., prehistoric influences, historical and present-day effects) and disturbance versus succession (Turner and Gardner, 2015). Disturbance is the key driver of spatial and temporal heterogeneity by altering the state and dynamics of a system (Johnson and Miyanishi, 2007). Accordingly, every landscape is affected on a wide range of scales by disturbance, which forces the creation of complex landscape mosaics (Turner and Gardner, 2015). This is particularly true for river and riparian landscapes whose dynamics are predominantly influenced by human disturbance in general (Aguiar and Ferreira, 2005) and the flow regime in particular (Tabacchi et al., 1998). Herein, a shifting mosaic (Stanford et al., 2005) is created by a non-equilibrium that depends on different scales from fluvial disturbance for sustaining functional processes (Ward et al., 2002) and is maintained by different metastable, oscillation and acyclic processes instilled by disturbance in natural rivers (Formann et al., 2013).

The combination of every of these concepts determines a particular river landscape with a dominant natural vegetation community (matrix) composed by groups (mosaics) of different homogeneous areas (patches) and contours (corridors) (Figure 7).
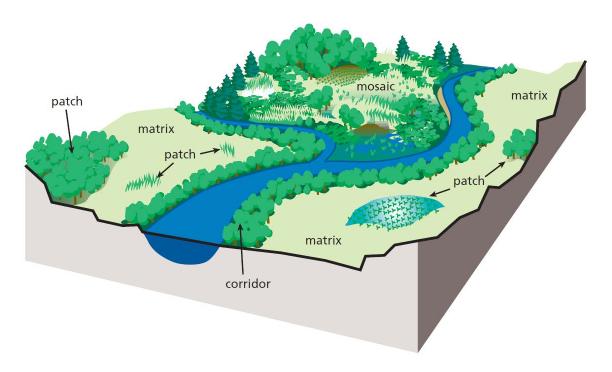
Figure 7. Spatial structure of the fluvial landscape (from: FISRWG, 1998).

The quantification of the riverscapes dynamics as an ecological response is critical to understand its functions (Malard et al., 2006) and, consequently, of great interest to landscape ecology (Fahrig, 2007). In fact, this quantification has proved to be an effective tool for guiding ecological restoration of riverscapes (Aguiar et al., 2011; Muñoz-Mas et al., 2017; Rivaes et al., 2017) as the underlying basic unit of the restoration is always the ecosystem (Clewell and Aronson, 2013). Ultimately, stream ecology as a landscape science stands as an improved attitude to create a truly holistic perspective of river structure and functioning (Fisher et al., 2001).

1.7. The peculiarity of Mediterranean ecosystems

There are only five Mediterranean climate regions (Med-regions) worldwide – Mediterranean Basin, coastal California, central Chile, South Africa's Cape region, and southwest and southern Australia – all lying approximately between 30º and 40º N and S the Equator (Figure 8).
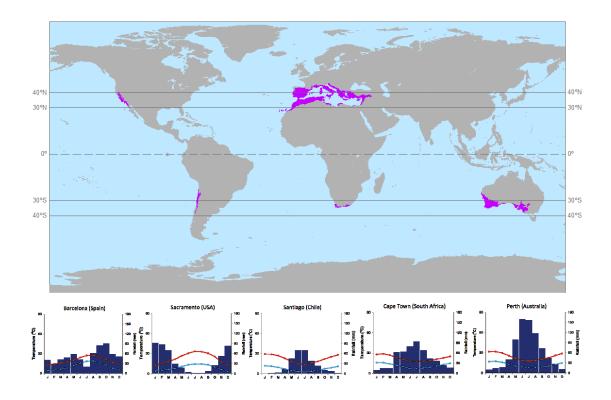
Figure 8. Mediterranean climate regions on Earth. Location and ombrothermic diagrams in the Mediterranean regions (adopted from: Dinerstein et al., 2017, and Batzer and Boix, 2016.

Together, these regions compose the Mediterranean biome, which harbors 20% of the world's total floristic richness in only 2% of the world's surface (Medail and Quezel, 1997; Olson and Dinerstein, 2002). Herein, plant diversity and density rivals with the tropical rain forests (Cowling et al., 1996). Moreover, vertebrate endemisms ascend to above 500, totaling approximately 2.1% of the total Earth's vertebrate species (Myers et al., 2000). Consequently, all the five Mediterranean regions are considered as hotspots for biodiversity and a global conservation priority (Myers et al., 2000). Notwithstanding, only less than 5% of this biome is formally protected by biodiversity protection reserves (Hoekstra et al., 2005).

The Med-regions present a very peculiar type of climate, mainly characterized by seasonality and variability of rainfall (Gasith and Resh, 1999). Seasonality is created by the intra-annual variability of rainfall, where a wet period of water surplus and seasonal flooding opposes another extremely hot and dry of hydric scarcity. The inter-annual variability is revealed by variations in the annual rainfall and thus by changes in magnitude of annual river flows and the frequency of floods. The annual precipitation

ranges from 300 to 900 mm, most of all during the mild winter period (Aschmann, 1973; Bonada and Resh, 2013). Consequently, the water balance is frequently negative, with the annual rainfall-to-potential evapotranspiration ratio ranging from 0.12 to 1 (Cobelas et al., 2005), which creates a deficit in water availability during the summer period.

Rivers in these regions are highly responsive to rainfall, and therefore so its flow regime is reflective of the rainfall pattern. Accordingly, during the winter period, Med-rivers are characterized by a low flow interspersed by floods, with different levels of flashiness according to the intensity of the rain events and the size and morphology of the watershed. During the summer, rivers present a very low flow or even null, when the wetted area is reduced to isolated pools or to the complete drying of the river channel. This circumstance, allied with a concomitant high human demand for water during summer, makes Med-rivers particularly vulnerable to water abstraction (Gasith and Resh, 1999).

This seasonality and inter-annual variability have obviously a great influence on the reliant ecosystems. The seasonal drying-up of Med-rivers is most likely the main large-scale driver of fish assemblages, independently from microhabitats and local biotic interactions (Magalhães et al., 2002). Herein, fish fauna presents low levels of species diversity but high level of endemism (Myers et al., 2000; Olson and Dinerstein, 2002; Abell et al., 2008). Furthermore, above 70% of the endemic species in Med-regions are already extinct or in the process of extinction (particularly in Spain, Portugal and Morocco, more than 20% of those are stated as *Vulnerable*), which places freshwater fishes in Med-regions among the most threatened in the world (Hermoso and Clavero, 2011).

Riparian vegetation in Med-rivers also present some particularities that differentiate this ecosystem from their counterparts in temperate or tropical regions. The riparian ecosystems of Med-regions are denser and more productive relative to the surrounding landscape (Barbour et al., 1993; Zaimes et al., 2010). Similarly, riparian ecosystems present greater plant diversity (Aguiar and Ferreira, 2005; Santos, 2010; Young-Mathews et al., 2010), more wildlife species (e.g., Patten, 1998; Matos et al., 2009; Seavy et al., 2009; Santos, 2010; da Silva et al., 2011; da Silva et al., 2015) and support greater biogeochemical fluxes and cycling rates (Lewis et al., 2009).

1.8. River damming

Dams are built to retain water for human needs, but also bring on environmental consequences. River damming interferes with the four-dimensional perspective of the lotic systems, obstructs the river continuum and affects the hierarchical organization of the riverine habitat. When this continuity is interrupted, the continuum processes and nutrient spirals are disrupted with a consequent longitudinal shift in the river's physical parameters and biological phenomena (Ward and Stanford, 1983).

Dams are responsible for modifying the timing and quantity of river flows (e.g., Williams and Wolman, 1984; Rood and Mahoney, 1990; Maheshwari et al., 1995; Ward and Stanford, 1995; Maingi and Marsh, 2002; FitzHugh and Vogel, 2010), influencing the temperature of the water (e.g., Nilsson and Renöfält, 2008; Bae et al., 2016), interfering with nutrient loads (e.g., Nilsson and Renöfält, 2008; Maavara et al., 2014; Maavara et al., 2015; Zmijewski and Wörman, 2015), trapping sediments (e.g., Kondolf, 1997; Walling and Fang, 2003; Kondolf et al., 2014; Asaeda et al., 2015), blocking fish migrations and disturbing aquatic communities (e.g., Freeman et al., 2003; Kocovsky et al., 2009; Freedman et al., 2014; Cooper et al., 2017), and affecting floodplain ecosystems (e.g., Kingsford, 2000; Nilsson and Svedmark, 2002; Shafroth et al., 2002; Naiman et al., 2005; Gordon and Meentemeyer, 2006; Braatne et al., 2007; Braatne et al., 2008; New and Xie, 2008; Benjankar, 2009; Egger et al., 2009a; Egger et al., 2009b; Rood et al., 2010; Benjankar et al., 2012; Johnson and Waller, 2012). Flow regime alterations can thus have numerous impacts, namely, geomorphological (Lloyd et al., 2004), ecological (Poff and Zimmerman, 2010) and biological (Stromberg et al., 2010a). Such impacts put at risk the provision of habitat for native species, the maintenance of deltas and the productivity of fisheries (Postel and Richter, 2003). As a result, river damming interferences instigate an ecological response from instream and riparian species, which is proportional to the alteration of the flow regime (Poff and Zimmerman, 2010). Depending on the severity of changes, thresholds may be crossed with unforeseeable consequences for mankind (Jenkins, 2003), given that ecosystems provide ecological services that are critical to the functioning of Earth's life-support system and give a very important contribution to human welfare (Costanza et al., 1997). In the end, modifications in river flow regimes are even expected to be intensified due to increased water withdrawals to satisfy human need (Alcamo et al., 2007a; Vörösmarty et al., 2010; Murray et al., 2012).

River restoration stands for the reestablishment of the structure and function of the riverine ecosystem in an attempt to return as closely as possible to the pre-disturbance condition and functions (FISRWG, 1998). This topic is of great importance in many countries and EU Member States are even being asked to actively improve the quality of their rivers and the resilience of ecosystems to climate change (CEC, 2009; Biesbroek et al., 2010), with proactive conservation efforts centered on maintaining ecosystem functioning, climate mitigation and adaptation (Palmer et al., 2008; Woodward et al., 2010).

For the restoration of an ecosystem, one needs an ecological reference where to aim the restoration measures, as well as for the post appraisal of the restoration project (Clewell and Aronson, 2013). The ecological reference target is usually defined by reference sites but it can also be determined by modeling techniques. The reference site is normally the best available example of the most undisturbed ecosystem condition within the same river typology, used to benchmark the restoration measures. In contrast, when these reference sites are not available, it is acceptable to call upon modeling techniques that calculate the expected reference condition in a certain location (Darby and Sear, 2008). Nevertheless, the reference systems may provide a naive confidence in the predictability of the restoration projects outcome (Hughes et al., 2005) and should nonetheless be supported by modeling techniques that are an asset by providing water managers with a prediction of the ecological implications resulting from the restoration actions (Stromberg et al., 2010a).

To restore a river, one can opt for three basic approaches: the nonintervention and undisturbed recovery, the partial intervention for assisted recovery, and the substantial intervention for managed recovery (FISRWG, 1998). Firstly, rivers have many times the capacity of recovering themselves and this must be regarded when establishing a restoration plan. Consequently, the nonintervention and undisturbed recovery consists of letting the natural course of the river follow its way. This is recommended for rivers that are already rapidly recovering without any further intervention. Partial intervention is indicated when a push is needed to facilitate natural processes in the already recover attempting river. When the recovery is not attainable for the ecosystem, a substantial intervention is needed for a managed recovery. The necessity to consider the spatiotemporal scales of the restoration projects, in which the larger and longer

durations of those the more valuable will be (Lake et al., 2007), is common to the three approaches.

However, in order to restore a river, the pressures to which the ecosystem is subdued should be firstly remove. Unfortunately, many times this is not possible as it goes against installed human water necessities. This is the case of the hydrologic alteration caused by river damming, which is the main driver of river degradation in Med-rivers (Hooke, 2006; Grantham et al., 2010; González del Tánago et al., 2012). Therefore, the possible restoration measures often include the implementation of human-assisted improvements such as the recovery of natural hydrologic, morphologic and ecological processes (Dufour and Piégay, 2009). Nevertheless, it is necessary to assess and fully understand the Driver-Pressure-State-Impact-Response framework (DIPSIR) (EEA, 2007) of each ecosystem. Modeling techniques can provide a valuable assistance on this assessment.

In this sense, one measure to restore rivers and riverine ecosystems is the release of environmental flows, a prominent feature of river management in many countries, used to reinstate the elements of the natural flow regime (Arthington, 2015). In this, riparian requirements must not be forgot as the restoration of riparian communities is an indispensable measure to recover the natural river processes and the most promising restoration action in many degraded rivers (Palmer et al., 2014).

1.10. Environmental flows

The conflicting urge to protect river environments while satisfying human water demand remains one of the most important challenges of our time (Nilsson and Berggren, 2000; Palmer, 2010). We need to retain water, but we must improve or maintain the ecological sustainability of our rivers, as well. 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). Furthermore, every dam, weir or levee change to a certain extent some component of the natural flow regime (e.g., Maheshwari et al., 1995; Nilsson and Berggren, 2000; FitzHugh and Vogel, 2010; Miller et al., 2013; Uddin et al., 2014b; a) and there are still hundreds of thousands of these structures worldwide without any implemented environmental flow regime (Richter and Thomas, 2007).

Accordingly, the importance of this subject is so outstanding that forced the scientific community to appeal to all governments and water-related institutions around the globe to engage in environmental flow restoration and maintenance in every river (Brisbane Declaration, 2007). Accordingly, the European Union (EU) developed a guidance document (EU, 2015) intended for a common implementation strategy (CIS) under the European Water Framework Directive (WFD). This document provides an EU definition of environmental flows and a common understanding on how to assess and determine those to be applied in the context of the next generation of river basin management plans. It is a great tool that provides a very comprehensive description of the existing knowledge about this topic while delineating innovative strategies to tread future paths regarding the protection of aquatic ecosystems in EU. Unfortunately, it still devotes very little attention to riparian ecosystems.

Environmental flows play an essential role in the conservation of freshwater ecosystems (Hughes and Rood, 2003; Arthington et al., 2006) and 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). Environmental flows management has been an ongoing scientific issue for the last decades (e.g., Arthington and Zalucki, 1998; Dyson et al., 2003; Hughes and Rood, 2003; King et al., 2003; Acreman and Dunbar, 2004; Rood et al., 2005; King and Brown, 2006; Poff et al., 2010), prompting the development of a large number of methods for its assessment. In general, environmental flow methods can be grouped in four main types, namely, hydrological, hydraulic rating, habitat simulation and holistic methodologies (Tharme, 2003). These can also be named, correspondingly, as lookup tables, desktop analysis, habitat modeling and functional analysis (Dyson et al., 2003). Additionally, two other categories can be considered: a fifth – combined methods – including environmental flow recommendations resulting from the combination of the previous methods, and a sixth – other – comprising other methods with less scope and not falling under the previous cases. The first three types compose the hard core of all methods, representing near 70% (30, 11 and 28%, respectively) of the global methodologies for environmental flow assessment (Arthington, 2012).

In the first group – hydrological methods – the approaches are grounded on the statistical analysis of hydrological information to determine environmental flow

recommendations. These were the earliest methods, created *ad hoc* in the United States during late 1940s to maintain economically important freshwater fisheries (Tharme, 2003). Such methods started to be determined as a fixed-percentage of the mean river flow but currently can include discharge variations to comprise the natural monthly, seasonal or other flow variabilities. Although these can be considered the most rudimentary methods, these are still the most commonly applied techniques on a global scale (Dyson et al., 2003; Tharme, 2003; Linnansaari et al., 2012). The highly frequently used Montana or Tennant method (Tennant, 1976), the Flow Duration Curve methods, the Range of Variability Approach (Richter et al., 1996), the Flow Translucency Approach (Gippel, 2001) or the Portuguese Alves and Bernardo (Alves and Bernardo, 2000) method (commonly known as INAG's method) are procedures included in this category.

The second group – hydraulic rating methods – comprehends the techniques that establish a relationship between the water discharge and the amount of habitat provided for aquatic species. These approaches started to be used from 1970s onwards, where the habitat is quantified by a hydraulic parameter (e.g. wetted perimeter, water depth, area of shallow streambed) used as surrogate for habitat features presumed to be limiting to the targeted biota. The method consists in measuring and interpolating the hydraulic parameters as a function of the discharge. Plotting the discharge *versus* hydraulic parameter usually draws an approximately log-shape curve with a breakpoint used as threshold for the minimum discharge to provide to the system without reducing drastically the amount of available habitat. The Wetted Perimeter (or Area) method (Gordon et al., 1992), the Toe-Width or Washington method (Swift, 1975; 1977) or the Flow Events method (Stewardson and Gippel, 2003) fall into this category.

The third group – habitat simulation methods – uses ecological information to support the prescription of environmental flows. These methods typically couple collected evidence about the habitat preferences of the species with the information on how the physical habitat changes with discharge. The outcome of this combination is likewise a curve relating the available habitat with the discharge and the interpretation of the results similar to the previous method. This group counts with the most scientifically and legally defensible methods for environmental flow assessment (Gore and Nestler, 1988; Dunbar et al., 1998; Tharme, 2003; Gordon et al., 2004). The Instream Flow

Incremental Methodology (IFIM: Bovee and Milhous, 1978; Bovee, 1982) or the Riverine Community Habitat Assessment and Restoration Concept (RCHARC; Nestler et al., 1995) are among the methodologies included in this group.

The fourth group – holistic methodologies – emerged during the 1990s in Australia and South Africa (Hirji and Davis, 2009), revolutionized the paradigm and way of thinking about environmental flows and are now clearly being increasingly applied worldwide. Holistic methodologies aim for a more comprehensive attitude meant to address the protection and sustainability of river systems as a whole (Arthington et al., 1992; King and Tharme, 1994; King and Louw, 1998) rather than only some targeted species. This include the combination of several ecosystem components like river geomorphology, ecohydrology, ecohydraulics, water quality, riparian and aquatic vegetation, aquatic fauna and other depending on the riverine ecosystem, or even estuarine water requirements. In these cases, the environmental flow regime is built based on the several water requirements of the considered ecosystem components. The Building Block Methodology (BBM: King and Louw, 1998), Ecological Limits of Hydrologic Alteration (ELOHA: Poff et al., 2010), Downstream Response to Imposed Flow Transformation (DRIFT: King et al., 2003) or the Portuguese AQUALOGUS-eFLOW (Godinho et al., 2014) are examples of holistic methods. Accordingly, it is now a scientific consensus that environmental flows must ideally be based on the ecological requirements of different biological communities (e.g., Poff et al., 1997; Arthington and Zalucki, 1998; Davis and Hirji, 2003; Dyson et al., 2003; Acreman et al., 2009; Acreman and Ferguson, 2010; Arthington et al., 2010; Arthington, 2012; Acreman et al., 2014) 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; Postel and Richter, 2003; Lytle and Poff, 2004).

1.11. The riparian ecosystem

The word "riparian" derives from the Latin term "*Riparius*", meaning what frequents or belongs to the bank of the river. The use of this expression in scientific literature appeared with a noteworthy use in the beginning of 1970's and have experienced an exponential increase over the last three decades (NRC, 2002). The riparian ecosystem is therefore a characteristic component of the river systems, found in flood-prone areas and bordering the water bodies. This ecotone performs the connection between

aquatic and terrestrial ecosystems (Junk et al., 1989; Naiman and Décamps, 1997; NRC, 2002) and, being a transitional ecosystem, presents high structural and compositional biodiversity and production (Naiman and Décamps, 1997; McClain et al., 2003). The riparian ecosystem is widely regarded as one of the most productive and diverse ecosystems on Earth (Tockner and Stanford, 2002). It provides housing to many of the most endangered biota in the world (Tockner and Stanford, 2002; Tockner et al., 2008), serves as a pathway for organic and inorganic material redistribution (Nilsson and Svedmark, 2002), and plays a decisive role on riverine integrity (Van Looy et al., 2013). Nevertheless, it is also one of the most imperiled ecosystems on Earth, mainly due to human activities (Welcomme, 1979; Sala et al., 2000).

The vegetation inhabiting this ecosystem – riparian or floodplain vegetation – is ruled mostly by the flow regime and its stream flow components (Karrenberg et al., 2002; Rood et al., 2003; Merritt et al., 2010). Indeed, the flow regime is considered the most important factor in the shaping of these plant communities (Poff et al., 1997; Richter et al., 1997; Riis and Biggs, 2003; Franklin et al., 2008). This circumstance makes riparian vegetation not only particularly vulnerable to flow regime changes (Bejarano et al., 2012; Perry et al., 2012), but also a proper environmental change indicator (Nilsson and Berggren, 2000; Benjankar et al., 2012; Rodríguez-González et al., 2014).

Riparian species can establish along rivers by seedlings or clonal processes (e.g., Rood et al., 1994; Herberg and Sarneel, 2017). Riparian colonization by seedling establishment is described as occurring according to the recruitment box model (Mahoney and Rood, 1998), where the establishment of the seedlings is limited to a window of opportunity determined by concomitant seed dispersal and particular river stage pattern. The seedlings survival depends on the existence of suitable soil moisture and must cope with the receding river stage that provides potential recruitment zones, but also determines a withdrawing of the water table levels and consequent soil drying. In order to survive, seedlings must be able to accompany the stage receding with a similar root growth ratio or will perish due to the hydric stress imposed by the disconnection between root system and the receding moisture zone. Riparian species have their seed release synchronized with the natural flow regime, coinciding with when this stage recede is normal to occur (Mahoney and Rood, 1998; Stella et al., 2006). Therefore, the level of conjugation between these factors determines the recruitment band of each year. Clonal establishment can follow an

analogous process, in which plant parts are washed away and start fixing roots when laid on the ground, or through the formation of ramets (Douhovnikoff et al., 2005).

After a successful establishment, vegetation starts to grow and developing wherever and as far as it is able to resist to the consequent river flow regime disturbances. The shaping effect of the flow regime on riparian vegetation is applied by a flood-pulsed disturbance (Junk et al., 1989), whose dynamics pattern substantiates the river processes that directly impact the riparian vegetation in its interactions with the surface and groundwater river flow (Camporeale and Ridolfi, 2006; Formann et al., 2013). The local hydraulic and hydrologic characteristics in the river determine the morphodynamic processes arising from flood-induced stress through vegetation entrainment, uprooting, stem breakage, burial or anoxia (e.g., Friedman and Auble, 1999; Bendix and Hupp, 2000; Edmaier et al., 2011; Bendix and Stella, 2013). On the other hand, the oscillation of the groundwater level (Jansson et al., 2007) determines a physiological effect of water stress control on plant growth and survival in different extents according to the species dependency on the connection of the root system with the groundwater table (Stromberg et al., 1996; Shafroth et al., 2000; Cooper et al., 2003; Baird et al., 2005; Lowry et al., 2011). The combination of all of these morphodynamic and physiological factors, affects the succession dynamics of riparian vegetation both physically and physiologically (Blom and Voesenek, 1996; Poff et al., 1997; Kozlowski, 2002; Džubáková et al., 2015). This combination is further intricate by the great inter- and intra-annual variability of the fluvial disturbance that affect the succession dynamics of riparian vegetation in a medium to long term (e.g., Junk et al., 1989; Poff et al., 1997; Richter et al., 1997; Toner and Keddy, 1997; Whited et al., 2007a; Mallik and Richardson, 2009; Greet et al., 2011a; Greet et al., 2011b; Johnson and Waller, 2012; Miller et al., 2013; Webb et al., 2013). During interflood periods, vegetation is still controlled by several physical and chemical factors including flow velocity and depth (Chambers et al., 1991; Riis and Biggs, 2003), light availability (Carr et al., 1997; Köhler et al., 2010), water temperature (Barko et al., 1986; Carr et al., 1997), riverbed grain size (Baattrup-Pedersen and Riis, 1999), and the nutrient content of the riverbed and water (Barko et al., 1986; Demars and Edwards, 2009). Altogether, these aspects determine a multiplicity of physical habitats that control the presence of vegetation flow response guilds (Merritt et al., 2010; Sarr et al., 2011; Bejarano et al., 2012), illustrated in discrete units of homogeneous vegetation that occur in different succession phases with diverse stands of specific ages, structural features and

species compositions (Stanford et al., 2005). Consequently, the riparian succession phase is a reliable indicator of the hydrologic disturbances, in which floods and droughts are the major stressors (Poff et al., 1997; Lytle and Poff, 2004; Stromberg and Boudell, 2013).

1.12. Drivers of riparian vegetation

The flow regime is considered the most important driver in the shaping of these plant communities (e.g. Poff et al., 1997; Richter et al., 1997; Karrenberg et al., 2002; Rood et al., 2003; Merritt et al., 2010). However, several other factors influence the particularly important interactions between flow regimes, geomorphology and riparian ecosystems (Gregory et al., 1991; Naiman et al., 2000; Fisher et al., 2007). These factors act as drivers of the riparian ecological dynamics in a hierarchy of spatial scales (Frissell et al., 1986; Allan et al., 1997). According to their level of influence, these can be classified into large- or small-scale drivers. Large-scale drivers such as precipitation, temperature and evaporation rates, characterize fluvial patterns at a watershed or river landscape dimension, influencing river flow regimes at a climatic, hydrologic and geomorphic level (Thorp et al., 2008). Small-scale drivers influence vegetation structure and composition at levels smaller than the reach scale, such as the meso- and microhabitats of the riverine communities. Examples of these drivers are variables such as flow shear stress, height above water table, flow depth or riverbed substrate.

1.13. Riparian vegetation interactions with the remaining ecosystems

Riparian areas perform important hydrologic, geomorphic and biological functions to a greater degree than upland areas, considering the proportional area they cover within a watershed (NRC, 2002). In fact, riparian vegetation and macrophytes in general interact biologically, physically and chemically with aquatic species (Gregory et al., 1991), therefore playing an essential role in aquatic habitat improvement (e.g., Gregory et al., 1991; Naiman and Décamps, 1997; Tabacchi et al., 2000; Pusey and Arthington, 2003; Broadmeadow and Nisbet, 2004; Naiman et al., 2005; Ghermandi et al., 2009; Dosskey et al., 2010; Salemi et al., 2012; Statzner, 2012; Wootton, 2012; Ryan et al.,

2013; Van Looy et al., 2013; Rood et al., 2015; Chase et al., 2016) and biological conservation (Broadmeadow and Nisbet, 2004; Van Looy et al., 2013).

There are several documented benefits to freshwater environments due to riparian vegetation (e.g., Naiman et al., 1993; Broadmeadow and Nisbet, 2004), as well as the opposite effect due to its deterioration (Casatti et al., 2012). The ecological condition of riverine vegetation is able to directly influence fish (Growns et al., 2003; Beltrão et al., 2009; dos Santos et al., 2015) and macroinvertebrate communities (Shilla and Shilla, 2012), as well as indirectly through the effect on food webs (Vannote et al., 1980; Baxter et al., 2005; Wootton, 2012). It also influences the aquatic habitat availability (Carpenter and Lodge, 1986), the stream bank stability (Berges, 2009; Hubble et al., 2010; Rood et al., 2015) and the provision of large woody debris (Fetherston et al., 1995). Riparian vegetation provides thermal regulation of rivers by overshadowing (Ryan et al., 2013), traps sediments and contaminants (Chase et al., 2016) and performs chemical uptake and cycling (Dosskey et al., 2010), protecting the water quality (Hey and Philippi, 1995; Anderson et al., 2006; Blackwell and Maltby, 2006; Dosskey et al., 2010).

The riparian ecosystem also provides goods and services that are directly valued by human societies. It influences hydrological processes (Tabacchi et al., 2000; Salemi et al., 2012) and the reduction of damages from floodwaters (Daily, 1997; Blackwell and Maltby, 2006; Dosskey et al., 2010). Additionally, it supplies suitable areas for bird watching, wildlife enjoyment and game hunting (Flather and Cordell, 1995; Holmes et al., 2004; Berges, 2009), and provides fish for food and recreation (Nehlsen et al., 1991; Naiman et al., 2000).

In contrast, this ecological balance can be called into question by human-driven disturbances and produce ecological and economic damages (Brundu, 2014). Disturbances such as the stabilization of the flow regime (Franklin et al., 2008; Riis and Biggs, 2003) or changes in nutrient concentrations on water runoff (Madsen et al., 2001; Jones et al., 2002; Mainstone and Parr, 2002; Köhler et al., 2010) can interfere with the development of vegetation, increasing the occurrence of exotic plant species (Bunn and Arthington, 2002; O'Hare et al., 2006; Greet et al., 2015), demoting species diversity and richness in riparian communities (Merritt and Bateman, 2012) or even allowing for the uncontrolled understory vegetation growth (Kamisako et al., 2007). This can also increase flood risk through higher flow resistance (Nikora et al., 2008;

Vereecken et al., 2006) and interfere with human water uses such as water abstraction, hydropower, recreation and river navigation (Gómez et al., 2013; Halstead et al., 2003). Thus, riparian vegetation management plays an essential role in water and landscape planning and must be an emerging environmental issue in order to ensure a successful river management and restoration.

2. Knowledge gaps motivating this thesis

Several limitations in river science prompted the research and genesis of this thesis. Despite all the existing knowledge about freshwater ecosystems, many gaps can still be found towards an adequate protection and comprehension of these ecosystems. Stella et al. (2013) reviewed hundreds of papers regarding riparian vegetation in Mediterranean rivers and observed a shortage of research in the western Mediterranean basin, with a need for a better understanding of the physical stressors that interact with increased water scarcity and climate variability. Besides, the specific paths by which the drivers affect the riparian landscape have so far been scarcely investigated, especially regarding the local disturbances at a reach scale. Another knowledge gap regarding Mediterranean ecosystems is the poor understanding about biologically-mediated ecological processes like the functioning of aquatic and riparian vegetation and their influence on river processes, or the impact of alien species on food webs and ecological processes (Cobelas et al., 2005).

In river restoration, much attention has been given over the years to channel morphology reconfiguration, but the implementation of restoration measures based on well-accepted theories on the riverine functional perspective is only in its beginnings (Palmer et al., 2014). Moreover, the majority of the restoration approaches of riparian ecosystems do not normally entail a coordination with flow regime and are centered only on very localized flood defense measures (Hughes and Rood, 2003).

On the other hand, environmental flow assessment still persists generally based on the requirements of a single biological group, mostly fish (Tharme, 2003; Acreman et al., 2009; Arthington, 2012), and lack the input from less typically monitored taxa (Gillespie et al., 2014), like riparian vegetation. Consequently, these environmental flow approaches still disregard the inter-annual flow variability that rules species with

longer lifecycles and miss the long-term perspective of the riverine ecosystem (Stromberg et al., 2010b), thus allowing for the degradation of riparian woodlands and subsequently the river system. The knowledge about the feedback effects of this degradation on aquatic communities is still very short and the efficiency of environmental flow regimes in the long term is practically unknown.

The cause-and-effect relationships between flow regime and riparian vegetation are particularly important in Mediterranean climates. Here, strong flow regulation by dams is widespread and the hydrologic-driven changes on vegetation are most likely more intense as flow regulation persists for longer river stretches downstream of dams (Bejarano and Sordo-Ward, 2011). The outgrowth of riverine vegetation is also particularly problematic in Med-rivers flowing through intensive agricultural watersheds where the vegetation removal by mechanical or chemical means are the paradigm of riverine vegetation management (Hussner et al., 2017; Madsen, 2000). In this context, the topic of using fluvial disturbance in favor of regulated river management has been rarely addressed. Little attention has so far been given to the possibility of managing channel encroachment and invasion through the establishment of artificial flows derived from reservoirs. Indeed, reservoir outflows were already prescribed for other purposes rather than the ecological maintenance of aquatic fauna. Reservoir outflows (in the form of flushing flows intended to mimic the effects of natural flows) for the removal of fine sediment and channel maintenance (Kondolf, 1998) or encompassing the scour of undesired vegetation (Milhous, 2012) were already recommended. However, the majority focused only on the sediment transport capacity of these flows, whereas the rejuvenation of the riparian patch mosaic was only occasionally considered. Moreover, many of these reservoir flows endeavoring to manage riverine vegetation were hardly ever tested before and therefore its efficiency remains uncertain. Thus, regardless of the development in disturbance research in rivers, there are still innumerous future research opportunities in this subject, such as the spatialization of disturbance and recovery, the disturbance regime modifications and responses, or simply the continuity of expanding the knowledge about disturbance (Stanley et al., 2010).

Given that flood cycles are paramount in influencing riparian forest patterns (Loučková, 2012), new tools are urgently needed to provide a long-term quantification of the predictable effects of stream hydrological re-setting on riparian dynamics (Wohl et al.,

2005; Stromberg et al., 2010b). Dozens of riparian vegetation models were reviewed by Merritt et al. (2010) with the objective of providing an overview of the most relevant hydrologic features influencing riparian vegetation patches. Those authors realized that, in general, the existing models treated only one or two species, did not specify time-scale of response and, although they could be very effective for specific river reaches or segments, could not be applied to other rivers or regions due to their specificity. Furthermore, some of the latest approaches to riparian vegetation modeling still miss a spatial output of the functional type dynamics, which is essential for predicting and managing riparian ecosystems as a whole (e.g., Primack, 2000; Auble et al., 2005; Camporeale and Ridolfi, 2006; Dixon and Turner, 2006; Orellana et al., 2012; Tealdi et al., 2013). So far, few studies have used a spatially detailed approach to riparian patches to examine the responses of riparian vegetation (e.g., Benjankar et al., 2012; Egger et al., 2012; García-Arias et al., 2013; Rivaes et al., 2013). Likewise, a valid assessment of the spatiotemporal shifts in different functional types of vegetation is essential to forecast feedbacks in stream flow changes and associated disturbance processes (Stromberg et al., 2010a).

Overall, the knowledge gaps that motivated the present research, regarding the overall ecosystem approach in Mediterranean fluvial ecology, were:

- Lack of research in the western Mediterranean basin;
- Low availability of models applicable to Mediterranean regions using landscape approaches;
- Scarce investigation about the specific paths by which drivers affect the riparian landscape;
- Lack of input from less typically monitored taxa in environmental flows;
- Disregard for linking processes between terrestrial and aquatic interfaces;
- Deficiency of research in the relations between riparian and aquatic species;
- Inexistence of long-term studies to support ecosystem and climate change understanding, as well as short-term processes due to intra and inter-annual environmental variability.

3. Tools used in this thesis

A successful river restoration and management requires a deep understanding of the functioning of river-related ecosystems, in order to be able to reduce the efficiency uncertainty of the adopted measures. This is where models can play an important role, as they can aid scientists improving knowledge about river ecosystems or provide water managers with the capacity of anticipating the ecological implications resulting from a certain restoration or management action. Furthermore, ecological modeling provides model-based testing faster and requiring less financial inputs than actual physical experiments (Perona et al., 2009; Schmolke et al., 2010). This is particularly important for the riparian ecosystem, as many of its changes do not become obvious in the short-term. Dynamic riparian vegetation models are interesting tools for these analyses because they consider ecologically relevant flow regime elements to simulate modifications of vegetation features. Additionally, these models can also provide a great assistance for the definition of the ecological reference to which point the restoration goals whenever this reference does not exist.

The floodplain vegetation model used in this thesis is the dynamic rule-based spatially distributed vegetation model *CASiMiR-vegetation* (Benjankar et al., 2011), which consists of several grid-based modules and functions with a Boolean logic relying on hard thresholds provided by users. This tool allows for the simulation of vegetation dynamics based on the relationship between ecologically relevant hydrological elements such as magnitude, frequency, rate of change, inter-annual variability and sequencing of flows (Poff et al., 1997), and riparian vegetation metrics which clearly reflect the vegetation responses to chronic hydrologic alteration, such as age distribution, composition, and cover (Merritt et al., 2010). In addition, it incorporates the historical patch dynamics into every simulation, together with observed data information and expert knowledge. This model is particularly interesting because, unlike earlier ones, it works at the response guild level (it is based on succession phases associated with different hydromorphological attributes), thus enabling scientists and managers to link the influencing parameters to the probable changes in vegetation type attributes, and allowing regional calibration and application in both individual streams and catchments (Merritt et al., 2010). *CASiMiR-vegetation* has already been successfully applied in other countries (Benjankar, 2009; Egger et al.,

2009a; Egger et al., 2009b; Egger et al., 2012), and has proven its usefulness in the assessment of riparian vegetation dynamics (Benjankar et al., 2010).

The *River2D* model (Steffler et al., 2002) is also used in this thesis to perform the necessary hydraulic modeling. This is a finite element model widely used in fluvial modeling studies for the assessment of habitat availability (Jalón and Gortázar, 2007; Boavida et al., 2011). It couples a two-dimensional 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. Moreover, it incorporates a bed resistance model and a transverse shear model. The *River2D* is based on the depth averaged *Saint-Venant* equations computing the depth and the discharge intensities in the x-y directions. It was developed for application in natural rivers and features wet-dry area solution capabilities by combining surface flow and groundwater flow equations to compute the free elevation above and below the ground (Ghanem et al., 1996). In the context of this thesis, *River2D* was used for the calculation of several hydraulic parameters and fish habitat availabilities that were used for the ecological modeling and performed analyzes.

4. Thesis objectives and structure

4.1. Thesis objectives

The focus of this thesis is mainly about the knowledge development concerning the close connections between flow regime changes, riparian vegetation dynamics, and related aquatic biota and fluvial processes (Figure 9). In the following chapters, one looks over the effects of flow regime changes, such as driven by climate change, river regulation or watershed management, into the structure and functioning of riparian vegetation, characterized by its succession dynamics and landscape features. Then, the influence of vegetation changes on the aquatic assemblages, sediment transport and water related issues are investigated as well. The modeling of riparian vegetation is performed mainly with the *CASiMiR-vegetation* model, which was calibrated and applied to Med-rivers for the first time.

Specifically, the objectives of this thesis are:

- Calibration and validation of a dynamic vegetation model based on the predictive relationship between riparian flow-response guilds and the river flow regime;

- Assessment of the main drivers of riparian vegetation's ecological succession and the evaluation of its relative influence towards the determination of riparian vegetation flow regime requirements;

- Development of an initial approach to riparian vegetation restoration measures by flow regime management;

- Establishment of local riparian reference conditions for environmental flows benchmarking in regulated rivers;

- Conception of a preliminary holistic frame for environmental flows by combination of both riparian and aquatic elements;

- Test and validation of a preliminary holistic frame in river reaches presenting different types of flow regulation;

- Prediction of structural and functional changes in river communities affected by long-term flow changes;

- Evaluation of the ecological feedbacks of riparian vegetation management on aquatic communities.
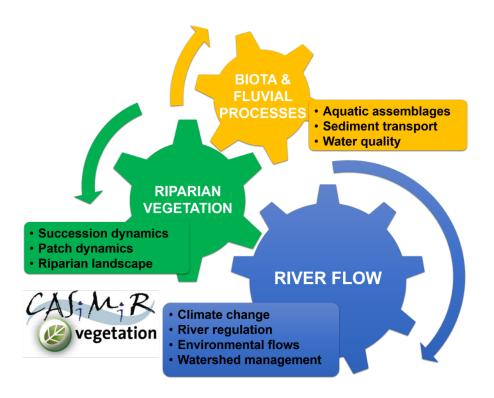
Figure 9. Thesis focus. Knowledge development concerning the ecological connections between flow regime, riparian vegetation, and aquatic biota and fluvial processes.

4.2. Thesis structure

The thesis comprises six sections with eight chapters, organized and planned in order to achieve the thesis objectives. Accordingly, Section I provides the essential background for a better understanding of the thesis aims and the investigations performed throughout the document. The following four sections – Sections II, III, IV and V – are devoted to present the research carried out to support the discussion and final conclusions of the thesis. Every of these sections focus on a particular theme with relevance for the proper consolidation of the thesis and can include one or two chapters according to their adequacy of the topics covered. Each chapter of these four sections corresponds to a published paper in international scientific journals with peer-review. These chapters' contents are identical to the published versions, but were formatted for the sake of thesis graphical uniformity. Finally, Section VI presents a general discussion of the results and the consequent conclusions of this thesis.

More in detail, the sections and chapters devoted to address and fulfil the thesis objectives are:

Section II – Large-scale drivers of riparian vegetation

The seasonality and variability of rainfall are accounted as major drivers of the river flow regime and consequently of riparian vegetation ecological succession. The influence of these drivers is assessed by a comparative analysis between the expected rainfall patterns ruled by future climate change scenarios and the current riparian landscapes governed by the actual river flow regime. This section includes two chapters. Chapter 2 presents for the first time a test of the model performance in a semi-arid system and of its ability to predict the riparian vegetation response to hydrological changes in a typical Med-river. It features also the prediction of the spatial and temporal changes of riparian succession phases in response to different flow regimes that were expected to result from climate-change scenarios. Chapter 3 reveals an assessment of the riparian vegetation structural changes caused by climate-changed flow regimes in different climatic and hydrogeomorphic contexts across Europe. By these means, it provides a broader view about the possible responses of riparian vegetation in different flow regimes and thus a better understanding of the specificities of riparian vegetation in Mediterranean ecosystems.

Section III – Small-scale drivers of riparian vegetation

The flow regime main drivers of the riparian patch mosaic are evaluated on a reach scale. Herein, two key drivers were considered to influence the riparian patch mosaic, namely, groundwater hydrology and morphodynamic disturbance. Chapter 4 investigates the effect of these fluvial disturbances on two central elements of landscape ecology, the location and shape of riparian vegetation patches, showing how the components of the flow regime affects the riparian landscape features, quantifying it and detailing those influences by succession phase.

Section IV – Riparian vegetation management

This section encompasses different studies that explore potential vegetation management options in regulated rivers, aiming at the restoration and conservation of

the riparian communities. In this, one attempts to establish a methodology to determine riparian vegetation requirements focusing on ensuring the long-term ecological quality of riparian vegetation in regulated rivers, as well as to propose restoration measures for instream and riparian vegetation by means of flow regime management. Two chapters substantiate these tasks. Chapter 5 presents simulations of the riparian vegetation response to different flow regimes with the intention of determining the flow requirements for the re-establishment of the riparian patch dynamics, and for the reduction of the flow regulation effects on riparian vegetation downstream of dams. An assessment of the potential damage of releasing sediment-deprived flushing flows on fluvial geomorphology is also analyzed in this section, as well as the adequacy of existing general environmental flow guidelines for riparian vegetation. Chapter 6 discloses a first time attempt to apply and validate hydraulic habitat modeling techniques to instream vegetation in order to determine a minimum annual flow with the ability to reduce the risk of channel encroachment and invasion by the alien species in a heavily regulated Mediterranean river.

Section V – Ecological feedbacks of riparian vegetation management

This section is dedicated to the appraisal of the vegetation management effects on aquatic communities. In chapter 7, the consequences of considering or not riparian vegetation requirements into environmental flows, concerning the habitat quality of fish species, are assessed. Consequently, the efficiency of environmental flow regimes is examined in the long-term perspective of the fluvial ecosystem. This task was approached from an ecohydraulic point of view in a groundbreaking way, with extreme potential for the prediction of expected long-term adjustments of river ecosystems.

SECTION II

LARGE-SCALE DRIVERS OF RIPARIAN VEGETATION

CHAPTER 2

Riparian vegetation responses to altered flow regimes driven by climate change in Mediterranean rivers

Paper published in *Ecohydrology*

Resumo

As regiões mediterrânicas caracterizam-se por uma grande variabilidade intra e interanual da precipitação e dos padrões de regime hidrológico associados. As previsões de alteração climática indicam que a precipitação média e temperatura anual irão aumentar, com uma concentração da precipitação e a ocorrência de períodos de seca mais prolongados e severos, motivando implicações profundas para os ecossistemas fluviais. O objetivo deste estudo foi prever a resposta da vegetação ripária mediterrânica a diferentes cenários de alteração climática, usando um modelo dinâmico de vegetação ripária que relaciona a dinâmica desta vegetação com o regime de escoamento. No local de estudo, os polígonos de vegetação mapeados mostraram ser significativamente distintos entre si, com a altitude, a altura acima do nível freático, a idade, e o diâmetro do caule como os fatores mais importantes na distinção das suas fases de sucessão. O modelo de vegetação ripária foi calibrado com uma boa força de concordância entre mapas observados e modelados. Os resultados do modelo em função do regime de caudais esperado sob o efeito das alterações climáticas demonstram que as áreas de vegetação não lenhosa esparsa se expandem para o exterior do leito do rio, com a consequente diminuição em área dos polígonos das fases mais jovens, enquanto os polígonos das fases de sucessão mais maduras se desenvolvem para o interior. Os resultados sugerem que as alterações climáticas extremas em rios mediterrânicos irão promover o desaparecimento dos estágios sucessionais pioneiros e jovens dos bosques ripários, transformando assim os esforços de conservação destes ecossistemas numa tarefa desafiante.

Palavras-chave: vegetação ripária, perturbação fluvial, regime hidrológico, modelo preditivo, sucessão de vegetação, alterações climáticas, CASiMiR-vegetation

Abstract

Mediterranean regions are characterized by a large intra and inter-annual variability in rainfall, and associated hydrological regime patterns. Predictions of changes in climate indicate that mean precipitation and annual temperature will both increase, with a concentration of precipitation and the existence of extended and harsher drought periods with profound implications for river ecosystems. Our aim in this study was to predict the response of Mediterranean riparian vegetation to different climate-change scenarios, using a dynamic riparian vegetation model that relates flow regime with riparian vegetation dynamics. In our case study, mapped riparian patches were significantly distinct in between, and altitude, height above water table, patch age, and stem diameter were the most important of the factors that distinguished succession phases. A floodplain vegetation model was calibrated and achieved a good strength of agreement between simulated and observed maps. Model results with the expected flow regime under the effect of climate change demonstrate that non-woody sparsely vegetated areas expand outwards and mature succession patches expand inwards, while pioneer and young riparian patches decrease in area. Our results suggest that extreme climatic change in Mediterranean rivers will promote the disappearance of the pioneer and young succession stages of riparian woodlands, thus making efforts to conserve these ecosystems a challenging task.

RIPARIAN VEGETATION RESPONSES TO ALTERED FLOW REGIMES DRIVEN BY CLIMATE CHANGE IN MEDITERRANEAN RIVERS

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Author contributions

Conceived and designed the experiments: RR PRG AA AP GE MTF. Performed the experiments: RR PRG AA. Analyzed the data: RR PRG AA AP. Contributed reagents/materials/analysis tools: RR GE AP. Wrote the paper: RR PRG AA AP GE MTF.

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1. Introduction

Since the beginning of the twentieth century, the average global atmospheric temperature has been rising and the speed with which it has been doing so has also increased (McCarthy et al., 2001). Global temperatures are expected to rise significantly in this century, even if greenhouse gas concentrations remain constant at year 2000 levels. Over the next hundred years, this will induce changes in the global climate system that will very probably be of a greater magnitude than those observed during the last hundred (Solomon et al., 2007). For Mediterranean regions, most Global and Regional Circulation Models that simulate the Earth's climate system predict an increase in mean annual temperature and a decrease in mean annual rainfall, with a concentration of precipitation in a short period of a few months each year and with extended and harsher droughts (Houghton et al., 2001).

The alteration of hydrological regimes may be as deleterious to ecosystems as temperature changes (Schröter et al., 2005), particularly in the case of river systems (Palmer et al., 2009). The principal attributes of the Mediterranean-type climate are seasonality and variability of rainfall, with seasonal flooding as a typical feature (Gasith and Resh, 1999). Inter-annual variability is represented by changes in the frequency of floods and variations in annual rainfall, while intra-annual variability (seasonality) is represented by periods of water surplus interspersed with hydric scarcity.

Even the most tenuous climatic changes can result in major alterations in the river flow regime, such as changes in the frequency of the recurrence of extreme floods (Sánchez et al., 2004). It is also expected that changes in natural flow regimes due to climate changes will be magnified by human efforts to satisfy water demand on the part of populations, such as water retention in reservoirs (McCarthy et al., 2001; Malmqvist and Rundle, 2002; Arthington et al., 2006). European Member States are nonetheless being asked to actively improve the resilience of ecosystems in the face of climate changes (CEC, 2009; Biesbroek et al., 2010), with proactive conservation efforts centered on maintaining ecosystem functioning, climate mitigation and adaptation (Palmer et al., 2008; Woodward et al., 2010).

Riparian vegetation is a characteristic component of river systems, and ensures a liaison between the river channel and the terrestrial ecosystems (Naiman and Décamps, 1997). The structure of this ecotonal zone and its lateral and vertical hydric connections are strongly related to the flow regime, which is the most important factor

in the shaping of these plant communities (Poff et al., 1997; Richter et al., 1997). The linkage between fluvial geomorphic processes and riparian vegetation dynamics creates a topographic diversity, soil moisture gradients and fluvial disturbance patches that characterize riparian ecosystems (Lite et al., 2005; Merritt et al., 2010). This makes these patch features valuable tools with which to assess the ecological integrity of river functioning. Climate-driven alterations in hydrologic regimes may thus have profound effects on riparian vegetation, yet few studies have addressed the magnitude of changes in rivers in semi-arid regions (Wilcox et al., 2003; Stromberg et al., 2010a).

Merritt et al. (2010) have reviewed dozens of riparian vegetation models with the objective of providing an overview of the most relevant hydrologic features influencing riparian vegetation patches. Those authors realized that, in general, the existing models treat only one or two species, do not specify time-scale of response and, although they can be very effective for specific river reaches or segments, cannot be applied to other rivers or regions due to their specificity.

The floodplain vegetation model used in this study is the dynamic rule-based spatially distributed vegetation model "CASiMiR–vegetation" (Benjankar et al., 2011), which consists of several grid-based modules and functions with a Boolean logic relying on hard thresholds provided by users. This tool permits the simulation of vegetation dynamics based on the relationship between ecologically relevant hydrological elements such as magnitude, frequency, rate of change, inter-annual variability and sequencing of flows (Poff et al., 1997), and riparian vegetation metrics which clearly reflect the vegetation's responses to chronic hydrologic alteration, such as age distribution, composition, and cover (Merritt et al., 2010). What is more, it incorporates the historical patch dynamics into every simulation, together with observed data information and expert knowledge.

This model is particularly interesting because, unlike earlier ones, it works at the response guild level (it is based on succession phases associated with different hydromorphological attributes), thus enabling scientists and managers to link the influencing parameters to the probable changes in vegetation type attributes, and allowing regional calibration and application in both individual streams and catchments (Merritt et al., 2010). Moreover, it uses 2D hydrodynamic modeling and is better at predicting the flow-shear stresses than 1D modeling (Horritt and Bates, 2002). *CASiMiR-vegetation* has already been successfully applied in other countries

(Benjankar, 2009; Egger et al., 2009a; Egger et al., 2009b; Egger et al., 2012), and has proven its usefulness (Benjankar et al., 2010) in the assessment of riparian vegetation dynamics.

In the present study, we tested the model's performance in a semi-arid system and its ability to predict the riparian vegetation response to hydrological changes in a typical Mediterranean river, both for the first time. Specifically, we aimed to predict the spatial and temporal changes in riparian succession phases in response to different flow regimes that we expected to result from climate-change scenarios.

2. Materials and methods

2.1. Study site selection

Three fundamental criteria were used for site selection: near-natural conditions in terms of the hydrologic regime (absence of flow regulation) and other human pressures; the naturalness of the riparian vegetation; and the existence of nearby hydrometric data. Southern Portuguese river basins were scanned to produce a list of potential study sites, using available land-use and flow-regime information, botanical literature and field surveys. From the list of possibilities, we chose a site in the Odelouca river basin (Figure 10a).

The Odelouca drainage basin is located in southern Portugal (between 37º10'49''N, 8º29'54''W and 37º26'33''N, 8º12'16''W), and has 511km^2 of drainage area and 92km of slow-running streams (Figure 10b). Average annual precipitation is approximately 750mm, mostly concentrated in a wet period from October to March and contrasting with a very dry one in the remaining months. These distinct periods characterize its typical Mediterranean hydrological regime, with winter flash peak floods contrasting with a dry bed and a few remaining pools in summer.

2.2. Field data

A field survey collected data about the habitat traits of riparian communities at the study site, and served to develop a vegetation map for use as a reference for model accuracy

evaluation. This task was carried out in the summer of 2009 and included a topographic survey and mapping of vegetation and habitat features. The topographic survey was performed using a Leica 500 GPS, composed of two double-frequency-to-real-time SR 530 RTK antennas L1 and L2 AT 502 (1cm error approximately), and an effort was made to record all changes in elevation greater than 20cm. The surveyed area was within the limits of the 100-year flood envelope and included aquatic, bank and floodplain zones (Figure 10c).

Aerial photo source: Portuguese Geographic Institute (IGP)
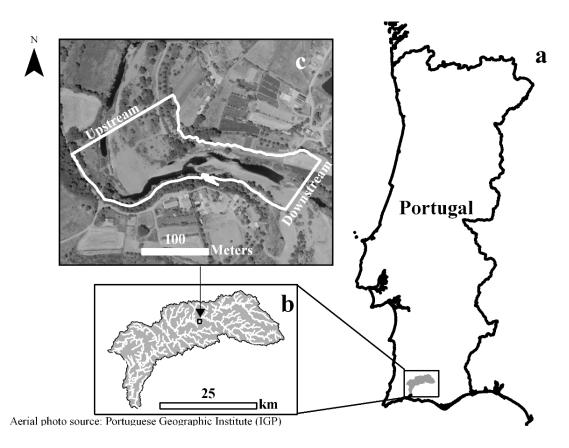
Figure 10. Location of study site and assessment area in the Odelouca river basin. Aerial photo source: Portuguese Geographic Institute (IGP).

Vegetation and habitat assessment considered homogeneous vegetation patches, each corresponding to one succession phase (adapted from Egger et al., 2009b and Naiman et al., 2005). From a previous appraisal of the study site, we identified five succession phases: Initial phase (IP), Pioneer phase (PP), Early Successional Woodland phase (ESWP), Established Forest phase (EFP), and Mature Forest phase (MFP). Each succession phase was distinguished by different soil and vegetation features.

In the sorting of the succession phases, the Initial phase was attributed to all patches dominated by open sand or gravel bars (less than 50% of vegetation cover), with an absence of woody potential arboreal species. Patches dominated by woody arboreal species recruitment, developing from either seedlings or propagules, were deemed Pioneer phase. The Early Successional Woodland phase classification was attributed to patches with a moderate standing biomass and well established individuals, dominated by pioneer species (like willows and tamarisks) that overwhelmed the remaining ones. Patches presenting high canopy cover and a well defined understory layer, dominated by facultative phreatophyte species like ash trees, were considered Established Forest phase. The latter patches were also characterized by the appearance of soil with an A-horizon, which was absent from the earlier succession phases. The Mature Forest phase was considered in patches that were also dominated by competitive woody and long-lived riparian species, but with the occurrence of terrestrial arboreal cork oak or holm oak species, which are typical of these climax Mediterranean landscapes.

The shape and position of each patch was georeferenced by walking along its limits with a submeter precision Trimble® GeoXT™ handheld GPS. GPS data was post-processed using Trimble® GPS Pathfinder® Office 3.00 to differentially correct, edit and export patch shape and georeferencing data.

Each patch was classified in terms of percentage of soil cover (percentage of soil in the patch not covered by vegetation) and soil substrate (percentage of rocks, boulders, stones, gravel, sand and fine elements), based on visual estimation. Vegetation patches were gathered by dominant substrate classes, from which we collected 13 samples for mechanical and organic matter content analysis. Samples were extracted from the first 40cm depth layer with a soil sampler, with retrieval of a composite sample of approximately 250cm^3. Soil texture was then classified by mechanical analysis, using the United States Department of Agriculture soil texture classification.

Vegetation feature inventories in each patch sought to characterize succession phases by vegetation attributes and included species identification, measurement of stem diameters, and aging. Woody species were considered phase indicators if they represented the dominant arboreal species in the patch in overall terms, and co-dominant if they had dominancy in the patch but were less present than phase indicator species. Species that were substantially present in the patch, generally characterizing

the succession phases in the lower vegetation layers, were assigned as companion species.

Stem diameter was obtained from three or four of the oldest individuals on each patch using a diameter caliper, by calculation of the mean of two crossed diameters for each individual. Stem diameters were recorded at breast-height level (DBH at ca. 1.3m) in specimens higher than 2m, near ground for smaller ones, and on the largest stem in the case of multi-stemmed individuals.

Patch aging was performed using dendrochronological methods. Three or four of the largest individuals were cored on each patch with a standard 5mm increment borer, taking two (three when needed) perpendicular cores at DBH in adult trees (Mäkinen and Vanninen, 1999). For individuals smaller than 5cm DBH, disks were obtained for age calculation purposes, and on multi-stemmed trees the cores/disks were taken from the largest stem. Additional trees were selected near the study site to complete the oldest girth classes for each species and make it possible to fit growth curves together with diameter data. The total number of samples taken for age determination was 140, corresponding to 81 trees (39 ash, 21 willow, and 21 tamarisk). Ring increments were measured to an accuracy of 0.01mm, using a RINNTECH® Lintab™ linear table and the TSAP-Win™ program, and age was attributed directly to all samples showing pith (59). For samples that didn't reach pith, age was estimated considering the sample diameter, the average bark width and the average ring width for the first 10 years of life in each species (obtained from ring increment measurement in samples showing pith) (Rodríguez-González et al., 2010).

2.3. Data analysis

Topographic data were used to create the Digital Elevation Model (DEM) of the site, for use in data-processing for hydrologic and vegetation modeling. The DEM (in raster format with 0.5m size cell) is a graphic representation of ground surface and was created in ESRI® ArcGis™ 9.2 from the elevation data gathered in the topographic survey. An observed vegetation map by succession phases was also created from patch records and their georeferentiation at the study site.

Relevant patch characteristics were summarized using Principal Component Analysis (PCA) in order to verify whether the model parameters were in fact the most responsive

ones in the Mediterranean case. This analysis was based on correlations and computed variances as SS/(N-1) and considered patch area, mean height to water table, patch age, mean stem diameter, and open soil as variables.

The assessed variables were tested using a Variance Analysis (ANOVA) to confirm significant differences between succession phases. The analyses were carried out with Statistica software (version 7.0) at a significance level of $\alpha=0.05$ and considered the succession phase as the categorical factor.

2.4. Hydrological data

Daily hydrological data for the Odelouca river were collected from the Portuguese Water Resources National Information System (SNIRH) database. A series of annual maximum discharges at the nearest gauging station was used to calculate the discharges for different recurrence periods at the gauging station section. The discharges at the study site were computed taking into account the area and the mean annual precipitation ratios between the drainage basins of the gauging station and of the study site. The computed discharges were used to assess the magnitude of the annual maximum discharges used in the vegetation model and to better understand the results provided by the vegetation model.

Water surface elevation for each flow in the study site was computed with HEC-RAS 4.0 (Brunner, 2008). 2D hydraulic modeling was performed using River2D 0.93 (Steffler and Blackburn, 2002) to obtain shear stress (shear stress can be defined as the stress applied by the flow parallel to the river bed; this stress is dependent on the hydraulic radius, energy grade line and water volumetric weight), and water-level data for minimum base flow.

2.5. Climate-change scenarios

The climate-change scenarios used in the present study were the predictions obtained from Global and Regional Circulation Models (GCMs and RCMs) for Portugal (Santos et al., 2002; Santos and Miranda, 2006) based on two different greenhouse-effect gas emission scenarios.

Of several models tested, GCM HadCM3 and RCM HadRM2 (created by the Hadley Centre for Climate Prediction and Research) produced the results that are most consistent with Portuguese historical observations (Santos et al., 2002). The RCM HadRM2 model results considered used the IPCC IS92a scenario, projecting an annual rate of increase in CO_2 of 1% (CO_2 concentration doubles in 95 years) with all other greenhouse gas concentrations accounted for in proportion to these CO_2 concentrations. The GCM HadCM3 model considered the B2 scenario from the Special Report on Emissions Scenarios (SRES), which considers a world focused on solving local social, economic and environmental issues, with a low rate of population growth until the late 21st century (Nakicenovik and Swart, 2000).

Both these models predict a tendency towards a concentration of runoff and precipitation in winter months, contrasting with a reduction in the rest of the year. HadCM3 – SRES B2 results for the South of Portugal show a winter runoff increase from about 30 to 50% by 2100, and HadRM2 – IS92a indicates a winter runoff increase from 60 to 130% for the Algarve region by the same date. Regarding the annual ground recharge, HadCM3 – SRES B2 expects a decrease varying from 0% to 10% in the South of Portugal, and HadRM2 – IS92a a decrease from 10 to 20% in the Algarve region, by 2100. This may represent a reduction of up to 1m in the water table according to HadCM3 – SRES B2, and up to 4m according to HadRM2 – IS92a (Santos et al., 2002; Santos and Miranda, 2006).

Based on results from the SIAM I and follow-up SIAM II research projects (available at http://siam.fc.ul.pt/siamI_pdf/ and http://siam.fc.ul.pt/siamII_pdf/), two future scenarios for hydrological regime change were created for inputting into the dynamic vegetation model: Scenario 1 – Optimistic; and Scenario 2 – Pessimistic. Two hydrologic regime databases were therefore created for use as drivers in the climate-change scenario modeling (Figure 11).

2.6.   *Vegetation modeling*

2.6.1. *CASiMiR-vegetation* model

Riparian vegetation was modeled using *CASiMiR-vegetation* (Benjankar et al., 2011), which is focused on floodplain physical processes and their influence on riparian

vegetation. The foundation of this model relies on the fluvial disturbance of the riparian patch mosaic caused by flow and flood pulsing (Junk et al., 1989; Walker et al., 1995; Tockner et al., 2000). To this end the model is structured in three main modules (Start, Dynamic and Visualization modules), which support users from the early creation of initial vegetation maps to the sequentially scenario-driven visualization of outputs (Figure 12).

The Start module recreates the natural potential patch disposal of the study site, according to topography and mean annual flow. It assigns succession phases to different heights above water level and provides an initial vegetation-patch layout if needed. However, this module's output is a static picture of the potential riparian vegetation at a study site and is intended to be used as just a starting point for the Dynamic module, if no previous vegetation assessment is available.

The Dynamic module simulates the influence of physical river processes on the survival and recruitment of riparian vegetation, and its output is a year-based temporal and spatial representation of the expected vegetation succession phases. It replicates the succession/retrogression of riparian vegetation, the shaping action of shear stress, flood duration and height over water table disturbances. Retrogression to the Initial phase occurs where shear stress or flood duration is higher than the vegetation resistance threshold. On the other hand, where vegetation is capable of resisting these stresses, succession takes place with consequent vegetation aging. Recruitment is also addressed in this module and is linked to water table elevation, as both recruitment and mortality by scour are related to flow regime (e.g., Johnson, 1994; Johnson, 2000) and water table elevations (Stromberg et al., 1996). This makes it possible to specify thresholds for any study site.
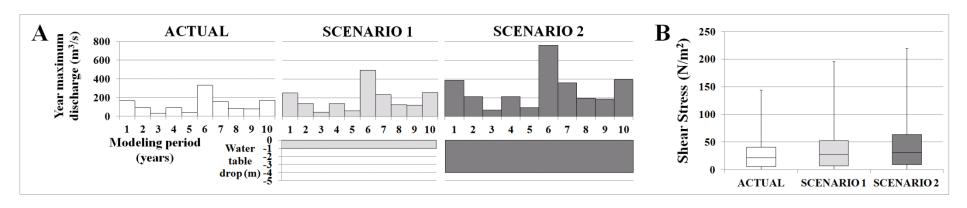
Figure 11. Hydrologic regime databases used to model riparian vegetation through the flow regime scenarios. A – values of maximum year discharge and water table declines for the flow regime scenarios over the ten year modeling periods. B – values of shear stress in the study site for each flow regime scenario, showing the median, first and third quartiles, and non-outlier extremes.
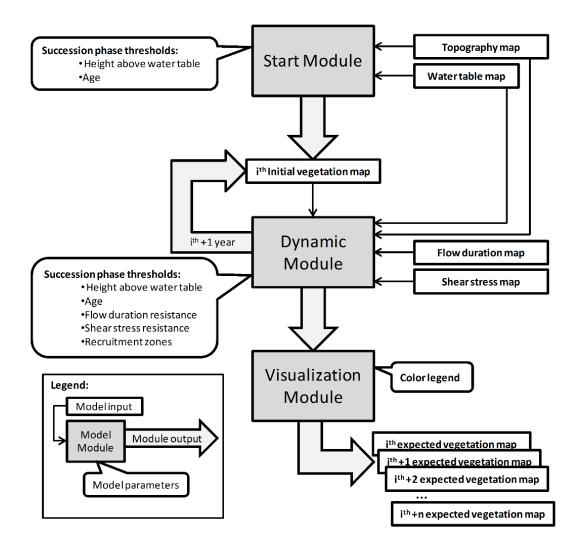
Figure 12. *CASiMiR-vegetation* model structure (adapted from Benjankar, 2009). Grey boxes stand for model modules, white sharp edge boxes represent the inputted or outputted model data and round corner callouts represent the thresholds definition driving model modules.

Riparian vegetation recruitment occurs in the Initial phase, wherever conditions for seeds to settle are met. The creation of suitable recruitment sites occurs in a band with adequate soil moisture, limited by the lowest level at which soil is considered to be usually too wet for establishment and the highest level at which it is too dry for seed settlement (Mahoney and Rood, 1998). These conditions are determined by model parameters, in terms of height over water table, which define recruitment areas in bank and floodplain zones above the scour disturbance zone. In the scour disturbance zone recruitment is unfeasible due to the water dragging influence caused by flow pulsing. Recruitment is thus settled for all the Initial phase areas of bank and floodplain zones, above the scour disturbance high-water mark and where soil moisture content is appropriate (Table 2).

Table 2. Calibrated parameters thresholds used in model computation. The parameters age, height above water table and shear stress resistance, characterize the biological traits of each succession phase, whereas scour disturbance, bank and floodplain recruitment, settle adequate conditions for the occurrence of recruitment in the modeled zones (in terms of height above water table level).

Model Parameters	Initial phase	Pioneer phase	Early Successional Woodland phase	Established Forest phase	Mature Forest phase
Age (years)	0 – 2	2 – 5	5 – 16	16 – 49	> 49
Height above water table (m)	0 – 0.363	0.363 – 0.683	0.683 – 2.778	2.778 – 3.405	> 3.405
Shear stress resistance (N/m^2)	30	30	50	300	300
Scour disturbance zone (m)	< 0.363				
Bank recruitment zone (m)	0.363 – 2.778				
Floodplain recruitment zone (m)	< 3.405				

In the Dynamic module, physical processes are modeled by zones (aquatic, bank, and floodplain), which are defined in advance for inputting into the model. The aquatic zone is usually defined as the area inundated by the base river flow. In this specific case, given that the river stops flowing in summer, the aquatic zone was attributed to the pools that remain when flow is null. The bank and floodplain zones were defined as the areas inundated by the 1-year and 100-year return period discharges respectively.

The Visualization module works as an aesthetic tool which enables the outputs to be visualized with a unified legend defined by the user.

2.6.2. Applying the *CASiMiR-vegetation* model to Mediterranean ecosystems

The *CASiMiR-vegetation* model was originally calibrated for and applied in a case study on the Kootenai in the United States (Benjankar et al., 2011), which is a representative example of a temperate river. Some adjustments were therefore needed in order to implement *CASiMiR-vegetation* in a Mediterranean case study. The major divergence between temperate and Mediterranean rivers is flow regime (Gasith and Resh, 1999). Case studies on the Kootenai and the Odelouca present very

different hydrological regimes, as well as unequal riparian communities with distinct species composition. The Kootenai is a large perennial river with a natural flow regime characterized by a snowmelt-driven peak in late spring (Hoffman et al., 2002), while the Odelouca is a small intermittent river with a typical Mediterranean regime that includes a peak winter flow marked by flash floods driven by rain events, intermittency due to river drying in summer, and a marked inter-annual variability. This fact leads to different patterns of vegetation response to fluvial disturbance – a difficulty that we expected this model to overcome, thanks to its conceptual structure. The main differences in fluvial disturbance and vegetation response between sites are described below, as is the procedure we adopted in order to address this concern.

Floods in the Kootenai and the Odelouca occur at different times of year and with different durations, and the latter's effects on vegetation also differ. Flood duration has minor impacts on Mediterranean flow regimes, as floods are brief, lasting less than a day and occurring outside the vegetation growing season, when species would be more sensitive to waterlogging (Yair and Kossovsky, 2002). Vegetation resistance to shear stress is also intrinsic to species and communities, which thus also react differently between sites. However, when correctly calibrated the model is expected to respond well to differences in discharge magnitude and flashiness between rivers. *CASiMiR-vegetation* was initially built for a purpose (the evaluation of the operational losses caused by the Libby Dam, USA, and the assessment of different restoration plan effects on riparian vegetation; for a better understanding see Benjankar (2009) other than the one we used it for. Our approach to the effects of climate change on riparian vegetation is nonetheless based on flow-regime modification, so despite different intensities and the ensuing vegetation response, the physical and physiological stressors remain the same (shear stress, height above water table, and flood duration), whatever the cause may be. Furthermore, using succession phases as modeling units means that riparian species dissemblance can be circumvented and allows the model to be applied to different locations (Merritt et al., 2010).

2.6.3. Modeling strategy

Ecological modeling can be approached through scenario development or forecasting, but preference for one rather than the other depends to a large extent on the length of the modeling period. The scenario development approach is especially useful for

environmental pattern predictions, whereas forecasting should be considered for short periods of time, due to the complexity of systems and their inbuilt uncertainties (Clark et al., 2001). In the latter approach, special attention should be paid to the time horizon, which is directly related to the level of uncertainty, because modeling errors are added over the course of the chosen modeling period. In the particular case of riparian vegetation, reaction to fluvial disturbance has been observed in immediately subsequent years (Scott et al., 1999; Amlin and Rood, 2003; Rood et al., 2003; Williams and Cooper, 2005; Hultine et al., 2010), as well as in reversible succession areas affected by flooding (Décamps et al., 1988). In addition, individuals and populations respond to the disturbance influence of river discharge on a yearly basis, with the decadal timeframe considered to be the appropriate spatio-temporal dimension for such analyses (Schumm and Lichty, 1965; Frissell et al., 1986; Thoms and Parsons, 2002; Thorp et al., 2008). We therefore committed to the use of climate-change flow scenarios and the forecasting of such modifications in the expected future landscapes over a decadal timeframe. In doing so, we also attempted to reduce the uncertainty inherent in modeling a period of several decades by going back to the scenario approach. At the same time, we were able to analyze the relevant fluvial disturbance of riparian vegetation in each scenario by forecasting the expected patch disposal for the respective disturbance time horizon.

2.6.4. Calibration and validation

In order to calibrate the model, each of the local existing succession phases had to be categorized by age, height above water table, and shear stress resistance. Age and height above water table parameters were defined from vegetation and habitat surveys of the study site, and were therefore calibrated by the establishment of thresholds based on *in situ* fieldwork. Flood duration impact does not exist in this specific case (as explained above, typical flash floods last for hours at most and always occur outside the vegetation growing period) and was thus not calibrated. For all simulations, flood duration was therefore considered null and no resistance thresholds were settled.

Calibration was achieved by running the model for a ten-year period (1999 to 2009), using the actual flow regime database. The initial vegetation map for 1999 that was used to begin modeling was obtained from the *CASiMiR-vegetation* Start module. In order to estimate shear stress parameters, during this procedure vegetation shear

stress resistance thresholds were tuned (see Wainwright and Mulligan, 2004 for a better understanding) by an iterative process of trial and error that sought to attain the best calibration outcome. To acknowledge the influence of this parameter on model output, we performed a basic sensitivity analysis using Cohen's Kappa (Cohen, 1960) statistic as an output metric. Varying these parameter thresholds in terms of maximum and minimum values resulted in a Kappa variation of no more than 0.04 units, which meant a 2% variation in output metric. The calibrated parameters used in model runs are presented in Table 2.

After calibration, model accuracy was evaluated by comparison between observed and expected 2009 vegetation maps. Maps were compared cell-by-cell and the resultant confusion matrix was analyzed with Cohen's Kappa, which has been already recognized as a helpful method for analyzing this model's accuracy (Benjankar et al., 2010).

The *CASiMiR-vegetation* model was validated through a 1995 aerial photography assessment of the study site. The model ran a ten-year period (1985 to 1995) hydrologic regime to produce the 1995 expected vegetation map, which was compared with the 1995 observed one. Once again, the Kappa statistic was used to appraise model accuracy.

After calibration and validation, the model was run with pre-created input information maps regarding the two different hydrologic regimes driven by the optimistic and pessimistic climate-change scenarios. For each scenario modeling a database containing all the input information was used to guide the model through the simulation process for the respective ten-year periods (2090 – 2100).

3. Results

Succession vegetation phases were sequentially disposed across a lateral gradient, from the deepest river area towards upland vegetation. This trend was also noted in different habitat variables related to the succession phase differentiation, such as particular indicator and co-dominant species (Table 3). In the case of the latter feature, the IP is characterized by the absence of phase indicator species, because no woody species were present in this phase. Having said this, the co-dominant species in IP

was *Rumex induratus* (Boiss et Reut.), as it was the most representative in those patches. In PP and ESWP, the phase indicator species were tamarisk and willow, with various herbs as co-dominant and companion species. The ash tree (*Fraxinus angustifolia* Vahl) was the phase indicator species in the EFP and MFP, together with the cork oak (*Quercus suber* L.) in the latter phase, with tamarisk (*Tamarix africana* Poiret) and the olive tree (*Olea europaea* L.) as co-dominant species, respectively.

Soil analysis showed the presence of two main textural soil types, with EFP significantly different from younger phases in terms of the percentage of fine substrate (ANOVA $F_{4,73}=5.3611$, $p=0.00077$). Bank zone patches were characterized by coarse elements, whereas floodplain patches had a finer texture, with the presence of an A-horizon soil.

Altitude, height above water table, age, and stem diameter were the most important of the factors that distinguished succession phases (loadings $>|0.75|$), with 69.63% of total variance explained by the first two axes of the Principal Component Analysis, thus endorsing the suitability of the model parameters for Mediterranean systems. Succession phases were significantly different for height above water table (ANOVA $F_{4,105800}=35231$, $p<0.0001$), except for IP, which presented a range that overlapped the PP and ESWP phases (Figure 11).

Turning to the succession phase aging, only EFP displayed a significant difference in age compared to the PP and ESWP phases (ANOVA $F_{4,59}=9.1435$, $p=0.00001$) (Figure 13). However, despite the non-significance of age displayed by PP and ESWP, at least 75% of the assessed patches of each succession phase ranged across different ages – i.e. the first three quartiles of PP patches were under 4 years old, and the last three quartiles of ESWP were more than 5 years old.

The vegetation map comparisons performed in calibration and validation returned a quadratic weighted Kappa of 0.63 for the former and 0.59 for latter, as a proportion of maximum possible values, given the observed marginal frequencies. In general, the model accuracy was therefore considered to have a good strength of agreement (Altman, 1991) between observed and modeled vegetation maps.

Table 3. Observed succession phases traits and habitat features at Odelouca study site – 2009 vegetation assessment. Mean ±Std. Dev. for quantitative characteristics.

Patch characteristics	Initial phase	Pioneer phase	Early Succession woodland phase	Established forest phase	Mature forest phase
Number of patches	15	12	32	12	11
Area (m²)	524.91 ±876.96	152.85 ±304.26	199.31 ±184.78	1328.46 ±747.55	1379.98 ±956.93
Altitude (m)	132.32 ±0.93	132.33 ±0.98	133.18 ±1.02	135.35 ±1.16	137.34 ±0.84
Height above water table (m)	1.21 ±0.92	0.53 ±0.28	1.69 ±1.15	3.14 ±0.53	3.86 ±1.04
Slope (%)	13.68 ±12.59	9.36 ±13.67	21.21 ±18.91	10.11 ±7.69	16.23 ±17.70
Age (years)	1.07 ±1.86	2.92 ±1.21	11.10 ±8.18	32.56 ±29.38	> 89*
Stem diameter (cm)	0.99 ±0.34	1.99 ±0.35	9.38 ±1.61	45.78 ±17.05	70**
Open soil (%)	48.05 ±31.78	45.13 ±30.90	13.59 ±18.78	1.92 ±6.64	3.11 ±5.33
Phase indicator species coverage (%)	-	32.63 ±23.60	68.75 ±39.60	46.92 ±47.86	35.82 ±38.90
Dominant texture	Coarse	Coarse	Coarse	Loamy sand	Loamy sand
Phase indicator species	-	Tamarix africana; Salix salviifolia	Tamarix africana; Salix salviifolia	Fraxinus angustifolia	Fraxinus angustifolia; Quercus sp.
Co-dominant species	Rumex induratus	Various herbs	-	Tamarix africana	Olea europaea
Companion species	-	-	Rubus sp.; various herbs	Rubus sp.; Olea europaea	-

* *Quercus* sp. age was not measured.
** Only one polygon assessed.
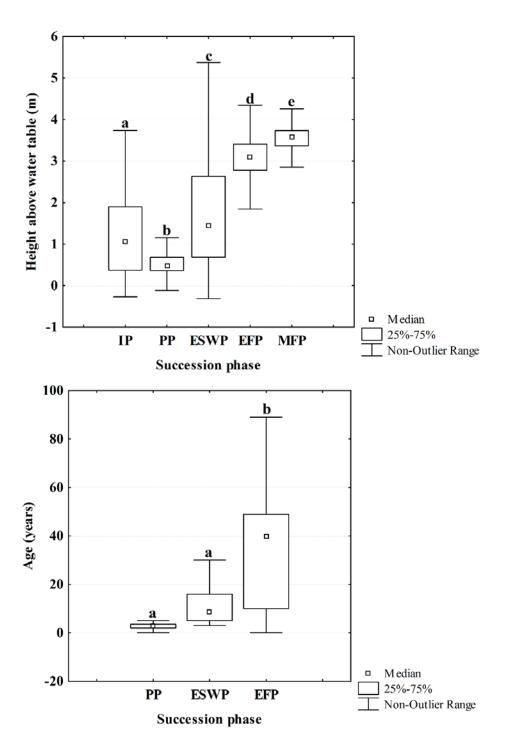
Figure 13. Height above water table and age discrimination of the succession phases observed in the Odelouca study site (vowels represent significantly different groups).

The differences between observed and expected maps never exceeded 12% of overall area; the greatest differences were found in the Initial and Pioneer phases, both of which were subject to a more erratic physical disturbance. MFP was recorded in 0.56% of the surveyed area, but was not expected by the Dynamic Floodplain Vegetation Model (Figure 14).

The vegetation modeling results for the actual scenario show that only the channel bed presented no woody vegetation. In this scenario, the first succession phase (IP) represented 23% of the total area, with PP plus ESWP vegetating in approximately 40% of it (18.6% for PP and 20.1% for ESWP). EFP was almost exclusively found in the floodplain zone and covered about 38% of the total area. Mature forest, with terrestrial species, was confined to the limits of the observed and expected actual scenario (Figure 14).
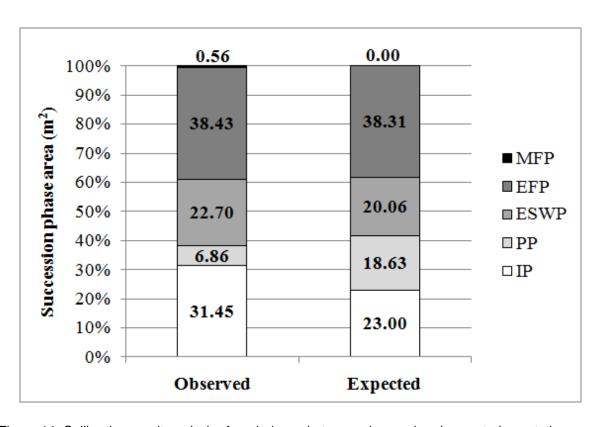
Figure 14. Calibration result analysis. Area balance between observed and expected vegetation maps (year 2009) with calibrated *CASiMiR-vegetation* model.

The modeling of the two climate-change scenarios shows a consistent decrease in area for the PP, ESWP and EFP phases (reaching the complete retrogression of PP and ESWP into IP in Scenario 2) and an increase in area for IP and MFP, when compared with the actual one (Figure 15). These results are attributed to the more intense floods and lower water tables expected for the climate-change scenarios, leading to an increase in IP caused by a washing away of the PP and ESWP phases, with the increase in hydric and shear disturbance not allowing ensuing recruitment.

MFP increased its area by replacing the previous succession phase in areas where hydric stress became the most important driver.

Comparing Scenario 1 with the actual one, we can see that IP becomes simultaneously present in bank and floodplain zones, thus explaining the increase of 16% in the total area covered by this succession phase (Figure 16). PP and ESWP together decreased by the same proportion (minus 9.7% and 5.7% of PP and ESWP respectively). EFP also decreased in about 13% of the total area, with an increase of the same magnitude for MFP, which can now be expected in the areas furthest from the main channel.

The comparison of Scenario 2 with the actual one reveals an increase of about 38% for IP, rising to 62% of total area due to the disappearance of PP and ESWP. In the floodplain zone EFP (12.6% of total area) also decreases and is replaced by MFP.
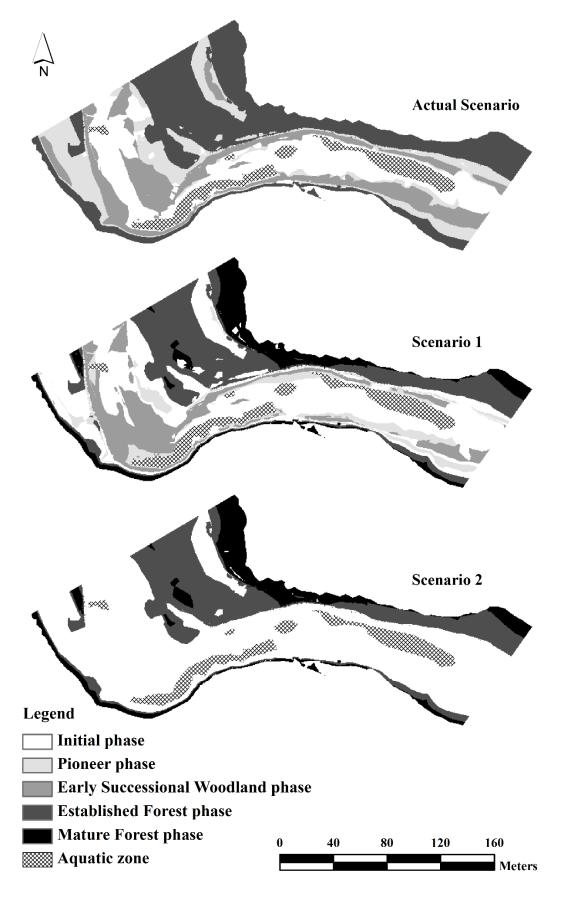
Figure 15. Expected vegetation maps in the actual and future climate change scenarios. Images show the spatial response of the expected vegetation succession phases to different flow regimes driven by climate change scenarios.
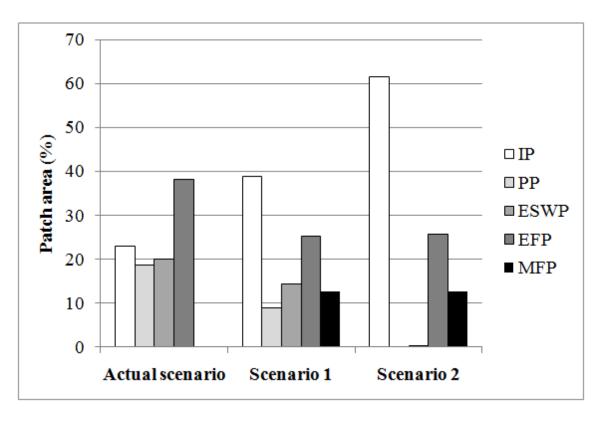
Figure 16. Expected patch area balance for the actual and climate changed scenarios. Columns represent the patch area of each succession phase in the three considered scenarios.

4. Discussion and conclusions

The present study was the first test of the performance of a riparian vegetation dynamic model (*CASiMiR-vegetation*) in a semi-arid basin, demonstrating its ability to predict the spatio-temporal changes in riparian vegetation guilds (i.e. succession phases) in response to climate-driven hydrological changes using a typical Mediterranean river as a case study. We were able to calibrate and validate a *CASiMiR-vegetation* model for such climate circumstances with substantial strengths of agreement and robustness in the face of uncertainty due to parameter estimation errors.

Our case study displayed a distinct spatial distribution of riparian succession phases from the inner channel outwards. The pioneer succession phase (PP) and the younger woodlands (ESWP) were characterized by patches of tamarisk and willow, while the outer areas of the fluvial corridor, corresponding to the established forest phase (EFP), were typically dominated by ash trees, whereas cork-oak woodlands corresponded to the uppermost terrestrial vegetation (MFP). The zonation can be related to the

biological traits of each indicator species. *Salix* spp. and *Tamarix* spp. are pioneer species that are adapted to disturbance. The former is a phreatophyte with restricted intervals of seed release and viability (Rood et al., 2010), and the latter a facultative phreatophyte species with a longer seed dispersal period and greater drought tolerance (Glenn and Nagler, 2005). On the other hand, the ash tree is a non-obligate phreatophyte that usually vegetates in silty soils, ranging from mesic to mesic and wet (Dufour and Piégay, 2008), and takes up water from unsaturated ground. Native tree species have developed strategies for overcoming the constraints imposed by the harsh Mediterranean environment and its prolonged summer drought, and it is thus possible to map a successional age sequence, which is adjusted to the flow regime in a way that is similar to the functioning of temperate rivers (Vadas and Sanger, 1997; Naiman et al., 2005; Willms et al., 2006). The rather abrupt change from successional riparian woodland (ESWP, EFP) to the mature phase (MFP), and especially the latter's characteristics (very old, low density single-stemmed trees), are probably influenced by human actions over a period of centuries, namely the transformation of the original mixed Mediterranean woodland into Montado-cork-oak agro-forestry-pastoral systems (Pereira and Pires da Fonseca, 2003). Pioneer (PP) and young-tree successional phases (ESWP) tend to be relatively larger in area than they are in temperate river zones (Friedman et al., 1996), due to the shear stress of Mediterranean winter flows and the ensuing increases in disturbance and the degree of difficulty facing the ability of riparian trees to persist over time.

Overall, the mapped riparian patches proved to be significantly distinct, with different positions in relation to the water table and with distinct ages. This was the case despite other environmental factors that compete with flow shaping, such as thermal stress and dryness, which can be present and have a less obvious shaping action on riparian patches (Katz et al., 2005). Because flow seems to be driving patch distinctiveness, it enables dynamic vegetation models like the one we used to predict patches, including for the Mediterranean flow regime, to which few have been successfully applied (Merritt et al., 2010).

In terms of model performance, *CASiMiR-vegetation* proved capable of successful implementation in relation to a Mediterranean riparian ecosystem, given that model accuracy assessment revealed adequate results in both calibration and validation. There are predictable changes in vegetation composition and age structure in both

river channels and floodplains. In the former, increased flood intensity and/or site drying ought to cause patch shifts in annual plants (retrogression to the initial phase), while in the latter, stand structure should change, with the replacement of woody species by others that are better adapted to the increased site dryness (Stromberg et al., 2010a). The results of this riparian vegetation model therefore appear to be correct, given the expected changes in riparian vegetation caused by a climate-change-driven modification of stream flow patterns.

However, at this stage such results should be seen from the point of view of ecosystem dynamics, inasmuch as significant differences in patch disposal can be found in between years and can prevent quantification approaches unless a stochastic methodology is considered. Performed sensitivity analysis showed that model uncertainty due to possible estimation errors in shear stress parameter thresholds was not alarming. Therefore, even admitting estimation errors in these parameters, results ought not to diverge too far from those achieved. However, we only analyzed the shear stress parameter, so a more thorough analysis is needed to assess all possible sources of model uncertainty.

Several models for assessing riparian vegetation ecosystems had been created prior to this one, and the differences between their structure and conceptualization means that comparing results with those of earlier models is a difficult task. Despite the shortcomings which Merritt et al. (2010) attributed to those models, many of them work as "black boxes", producing results based on statistical methods without fully detailing the physical and biological processes behind ecosystem dynamics. Furthermore, almost all the models approach flow regime disturbance through regulation and disregard climate-change-driven flow regimes, which present opposite disturbances in Mediterranean climates. Even so, similar patterns have been expected for specific equivalent disturbances tested in other models, such as a decrease in obligate phreatophyte species in flow regimes that have departed from their natural condition (Lytle and Merritt, 2004), high mortality rates due to falls in the water table of at least 1m (Scott et al., 1999), expected high sensibility of recruitment to climate-changed flow regimes (Dixon and Turner, 2006), and similar pattern in European boreal rivers (Ström et al., 2012) or in the large river Elbe (de Kok and Booij, 2009) when subjected to climate-change-driven flow regimes.

In addition to the consentaneous results, the adopted approach goes a little further and determines riparian vegetation dynamics in a spatially explicit way. The ability to predict riparian patches is an important dimension for management purposes, including the definition of flow regimes, temporal backwards or forwards-descriptive riparian scenarios, environmental directives for reservoir outflow management, river restoration planning, and ecological quality assessments. This predictive capability permits a very important aspect of river management – the long-term perspective (Stromberg et al., 2010b), the need for which is highlighted by the natural flow variation during flood events in different periods (year or multi-year), which are particularly relevant and need to be considered in order to achieve different ecological objectives (Rood et al., 2005).

Overall, under climate-change scenarios, non-woody sparsely vegetated areas expand outwards and mature succession patches expand inwards, while pioneer areas and young riparian patches decrease in area and are replaced by herbaceous vegetation. The magnitude of the winter discharges predicted in Scenario 2 (coupled with the lowering of the water table during the extended summer and the shortened spring) drastically affect tree recruitment and enlarge the initial phase area to over 50% of the studied area. This occurs because seed dispersal and seedling establishment lose their synchrony with hydrological processes (Mahoney and Rood, 1998; Stella et al., 2006) and there is therefore a lower survival rate and less time for consolidation into riparian woodland stands (Hargreaves and Allen, 2003). On the other hand, climax (upland) forest at the limits of the river corridor expands into formerly riparian floodplain patches, inasmuch as it is less affected by high winter flows or soil dryness in the floodplain zones.

In the long run and for the more pessimistic scenario, our results suggest that extreme climatic change will promote the disappearance of pioneer or young succession stages of the riparian woodlands in Mediterranean rivers. Concern on the part of society ought therefore to motivate discussion about possible courses of action for riparian conservation, such as setting restoration goals in order to maintain sustainable river ecosystems while accounting for human services, rather than attempting to regain historical landscapes (Dufour and Piégay, 2009). The second generation of basin plans *senso* Water Framework Directive ought to be climate-proof, in the sense that it should incorporate forms of adaption to climate change, protecting the aquatic ecosystem

while guaranteeing that human uses will not conflict with ecological quality standards (EEA, 2009).

This could well be an unfeasible scenario if the capacity of Mediterranean riparian woods to adapt to such drastic changes proves limited. However, the ability of riparian Mediterranean vegetation to develop functional and structural adaptive strategies for climate change at the proper pace remains an open question. Feedbacks between riparian vegetation and stream low-flow changes may become homeostatic if there are plant ecophysiological adaptations that ameliorate declines in base flows arising from increased aridity (Stromberg et al., 2010a). Moreover, other anthropic or natural-driven changes on different scales may also interact with the climate, thus increasing uncertainty about how vegetation will evolve. Overall, further developments in methods and simulation tools, like dynamic vegetation models, are required so that society can better assess the measures to be taken in order to restore and preserve riparian ecosystems.

Acknowledgements

CHAPTER 3

Modeling the evolution of riparian woodlands facing climate change in three European rivers with contrasting flow regimes

Resumo

As previsões dos modelos climáticos de circulação global indicam uma modificação dos padrões de temperatura e precipitação em todo o mundo. Tal fenómeno tornar-se-á particularmente evidente na Europa onde as alterações climáticas poderão ser mais severas do que a alteração média ao nível global. Consequentemente, prevê-se a alteração dos regimes de caudal dos rios com impactos resultantes nos ecossistemas aquáticos e ripários. Os bosques ripários encontram-se entre os ecossistemas mais ameaçados da terra e providenciam serviços vitais para os ecossistemas interconectados e sociedade humana. Contudo, foram objeto ainda de poucos estudos delineados com o intuito de quantificar a reação espaciotemporal destes ecossistemas aos regimes de caudais induzidos pelas alterações climáticas. O objetivo deste estudo foi analisar os efeitos de regimes hidrológicos induzidos por alterações climáticas na vegetação ripária de três regimes hidrológicos ocorrentes na europa. Os casos de estudo foram selecionados à luz dos modos mais comuns de alimentação da bacia que ocorrem transversalmente às regiões europeias, com o objetivo de avaliar as alterações esperadas nos elementos ripários dos sistemas fluviais devido às alterações climáticas. A modelação da vegetação ripária foi realizada com recurso ao modelo *CASiMiR-vegetation*, que baseia a sua computação na perturbação fluvial do mosaico ripário. Os resultados da modelação mostram que os bosques ripários poderão experimentar não só pelo menos alterações moderadas para todos os regimes hidrológicos, mas também alguns ajustamentos dramáticos em áreas específicas de estágios particulares de desenvolvimento da vegetação. Há circunstâncias nas quais a aniquilação completa da vegetação é uma hipótese plausível. Os regimes de caudais pluviais, tais como os dos rios do sul da europa, são os que provavelmente experienciarão alterações mais pronunciadas. Para além disso, independentemente do regime hidrológico, os indivíduos mais jovens e com maior dependência da água freática serão provavelmente os mais afetados pelas alterações climáticas.

Palavras-chave: vegetação ripária, alterações climáticas, modelação de vegetação ripária, perturbação fluvial, Europa

Abstract

Global circulation models forecasts indicate a future temperature and rainfall pattern modification worldwide. Such phenomena will become particularly evident in Europe where climate modifications could be more severe than the average change at the global level. As such, river flow regimes are expected to change, with resultant impacts on aquatic and riparian ecosystems. Riparian woodlands are among the most endangered ecosystems on earth and provide vital services to interconnected ecosystems and human societies. However, they have not been the object of many studies designed to spatially and temporally quantify how these ecosystems will react to climate change-induced flow regimes. Our goal was to assess the effects of climate-changed flow regimes on the existing riparian vegetation of three different European flow regimes. Cases studies were selected in the light of the most common watershed alimentation modes occurring across European regions, with the objective of appraising expected alterations in the riparian elements of fluvial systems due to climate change. Riparian vegetation modeling was performed using the *CASiMiR-vegetation* model, which bases its computation on the fluvial disturbance of the riparian patch mosaic. Modeling results show that riparian woodlands may undergo not only at least moderate changes for all flow regimes, but also some dramatic adjustments in specific areas of particular vegetation development stages. There are circumstances in which complete annihilation is feasible. Pluvial flow regimes, like the ones in southern European rivers, are those likely to experience more pronounced changes. Furthermore, regardless of the flow regime, younger and more water-dependent individuals are expected to be the most affected by climate change.

Keywords: floodplain vegetation, climate change, riparian vegetation modeling, fluvial disturbance, Europe

MODELING THE EVOLUTION OF RIPARIAN WOODLANDS FACING CLIMATE CHANGE IN THREE EUROPEAN RIVERS WITH CONTRASTING FLOW REGIMES

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Conceived and designed the experiments: RR PR MTF AP EP GE AG FF. Performed the experiments: RR EP AG. Analyzed the data: RR PR MTF AP EP GE AG FF. Contributed reagents/materials/analysis tools: RR PR MTF AP EP GE AG FF. Wrote the paper: RR PR MTF AP EP GE AG FF.

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1. Introduction

For decades scientists have been raising awareness about ongoing global climate change brought about by anthropic greenhouse gas (GHG) emissions into the atmosphere (e.g., Benton, 1970; Lovelock, 1971; Bach, 1976; Hansen et al., 1981). While at first it was possible to raise doubts in relation to the alleged global climate change process, the development and continued improvement of global circulation models (GCM) has allowed the scientific community to project with a high level of confidence that global mean surface temperature will increase over the course of the 21st century (IPCC, 2008). What is more, this trend will be followed by an increase in global averaged mean water vapor, evaporation and precipitation (Meehl et al., 2007). In Europe, regional circulation models (RCM) forecast climate warming above the projected global mean temperature rise, with precipitation pursuing contrasting tendencies according to region and season (Alcamo et al., 2007b). In Northern Europe, annual rainfall is expected to increase, while the opposite trend is expected for southern Mediterranean areas (Christensen and Christensen, 2007). Nevertheless, seasonal precipitation estimates in these regions are not straightforward. If winter precipitation in northern and central Europe is very likely to rise, in southern Europe there are some uncertainties, with different rainfall projections depending on the emissions scenario. On the other hand, it is consensual that summer rainfall will decrease all over Europe, and the same is true for snow, which is predicted to decrease throughout this continent (Christensen et al., 2007).

Such meteorological changes will significantly affect European river flow regimes, essentially through more pronounced low flow magnitudes in the Mediterranean climate zone and major modifications in high flow magnitudes in snow climates (Schneider et al., 2013). In summer, higher temperatures and evaporation rates, combined in a number of cases with less precipitation, will reduce runoff in many European regions (Nijssen et al., 2001; Serrat-Capdevila et al., 2007; Serrat-Capdevila et al., 2011). Even in nival or glacier-affected basins, runoff is expected to decrease due to a decline in melt water (Verzano and Menzel, 2007), leading to important reductions in floodplain inundations in the summer season. In contrast, higher runoff values in the wet season can enhance the risk of flooding caused by increased heavy rain events in a Mediterranean climate, or sleet (commonly known as "rain on snow events") in snow ones (IPCC, 2008). This will be further aggravated by the likelihood

that modifications in river flow regimes and their associated ecosystems will be amplified by future climate change interactions with anthropogenic pressures, such as increased water withdrawals to satisfy human needs (Alcamo et al., 2007a; Murray et al., 2012).

Rivers have a natural flow regime, on the basis of which aquatic and riparian communities have evolved in reliance on the ecological integrity of their ecosystems (Poff et al., 1997). Flow regime alterations can thus have numerous impacts – geomorphological (Lloyd et al., 2004), ecological (Poff and Zimmerman, 2010) and biological (Stromberg et al., 2010a) – on those communities. Depending on the severity of changes, it may be that thresholds will eventually be crossed with unforeseeable consequences for mankind (Jenkins, 2003), given that ecosystems provide ecological services that are critical to the functioning of Earth's life-support system and give a very important contribution to human welfare (Costanza et al., 1997).

Riparian ecosystems are particularly vulnerable to flow regime changes (Perry et al., 2012), since they are governed mostly by that regime and its stream flow components (Karrenberg et al., 2002; Rood et al., 2003; Merritt et al., 2010). Riparia forms a transitional boundary that connects aquatic and terrestrial communities (Junk et al., 1989; Naiman and Décamps, 1997; NRC, 2002), consequently presenting high biodiversity and production (Naiman and Décamps, 1997; McClain et al., 2003) while simultaneously harboring the most endangered ecosystems on earth (Tockner and Stanford, 2002; Tockner et al., 2008). Additionally, riparian areas perform important hydrologic, geomorphic and biological functions to a greater degree than upland areas, considering the proportional area they cover within a watershed (NRC, 2002). Indeed, researchers have documented several benefits to freshwater environment occasioned by the presence of riparian vegetation (e.g., Naiman et al., 1993; Broadmeadow and Nisbet, 2004), as well as evidence of the effects of its deterioration on instream species (Casatti et al., 2012). Riparian ecosystems also provide goods and services that are directly valued by human societies, such as reductions in damage from floodwaters (Daily, 1997; Blackwell and Maltby, 2006), supplying suitable areas for bird watching, wildlife enjoyment and game hunting (Flather and Cordell, 1995; Holmes et al., 2004; Berges, 2009), or providing fish for food and recreation (Nehlsen et al., 1991; Naiman et al., 2000). Thus, if decision-makers want to ensure that river restoration and administration produce successful results, they must consider riparian management to

be an emerging environmental issue that plays an essential role in water and landscape planning.

Given that flood cycles are paramount in influencing riparian forest patterns (Loučková, 2012), new tools are urgently needed to provide a long-term quantification of the predictable effects of stream hydrological re-setting on riparian dynamics (Wohl et al., 2005; Stromberg et al., 2010b). Also, a valid assessment of spatiotemporal shifts in different functional types of vegetation might become essential to forecast feedbacks in stream flow changes and associated disturbance processes (Stromberg et al., 2010a). Although valuable, some of the latest approaches to riparian vegetation modeling still lack a spatial output of the functional type dynamics, which is essential for predicting and managing riparian ecosystems as a whole (e.g., Primack, 2000; Auble et al., 2005; Camporeale and Ridolfi, 2006; Dixon and Turner, 2006; Orellana et al., 2012; Tealdi et al., 2013).

In the present paper we endeavor to assess riparian vegetation structural changes caused by climate-changed flow regimes in different climatic and hydrogeomorphic contexts across Europe, as well as to consider responses to emerging topics that are yet insufficiently studied in fluvial ecosystems (see Winemiller et al. 2010 for a better understanding), particularly with regard to riparian vegetation.

Preliminary results addressing such issues have been presented by the authors (Politti et al., 2011; Rivaes et al., 2012), but not as comprehensively and using old-fashioned scenarios in some cases. The present work goes beyond the scope of those earlier results, inasmuch as it further analyzes riparian patch amendments in accordance with climate-driven hydrologic changes. Moreover, this is the first time that a joint effort to ascertain the spatiotemporal response of riparian ecosystems to climate-changed flow regimes, considering the latest climate change scenarios with available regional hydrologic forecasts and on a European scale basis, has been made.

2. Methods

2.1. Ethics Statement

This study was conducted on hydric public domain locations at the three considered countries. No specific permits were necessary for the described field studies as the performed observational assessments do not qualify as a procedure requiring a license under the national legislation of any of the mentioned countries. Field studies didn't involve elimination or removal of any endangered or protected species.

2.2. Study site selection

Three study sites were selected with a view to encompass the principal watershed alimentation modes occurring across Europe. Although this was the primary criterion, we also attempted to consider an existing climatic gradient, determined by variables such as latitude, altitude or air temperature. Study sites (river reaches) were thus located in different countries with diverse climates and flow regimes (by both main water alimentation mode and transient pattern of discharge), namely Austria, Portugal and Spain (Figure 17).
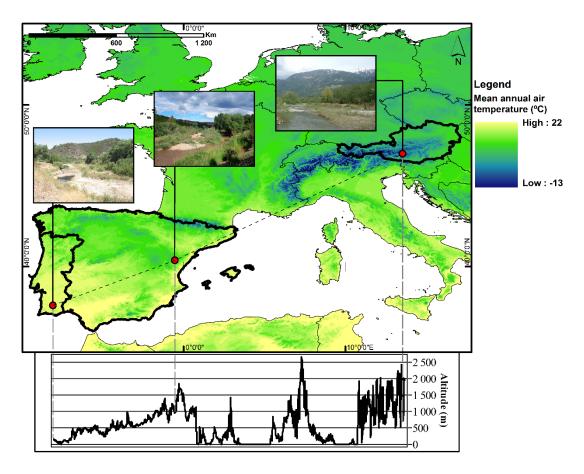
Figure 17. Study site location showing the spatial variation in mean annual air temperature and an altitude profile across the three study sites (Digital Elevation Model and Mean annual air temperature data source: EDIT Geoplatform, [January, 2013], (CC BY-NC-SA 2.5 ES), http://edit.csic.es/).

2.2.1. Kleblach reach (Drau River, Austria)

The Austrian case study is representative of the central Europe flow regimes, where maximum flows occur in spring and are attributable to snow-melt and glacial thaw. The study site is located at an altitude of approximately 570 meters in the upper river Drau, next to the village of Kleblach. Study site length is about 700 meters, and bank protection had been removed during an earlier river restoration project. Riparian vegetation comprises several species, most importantly including purple reed grass [*Calamagrostis pseudophragmites* (Haller f.) Koeler], German tamarisk [*Myricaria germanica* (L.) Desv], several willow species (*Salix triandra* L., *Salix purpurea* L., *Salix eleagnos* Scop. and *Salix alba* L.), grey alder [*Alnus incana* (L.) Moench] and European ash (*Fraxinus excelsior* L.). The river flow regime typifies a permanent temperate river, characterized by a mixed nivo-glacial regime (Mader et al., 1996) with significant flow (mean discharge between 1951 and 2008 equal to 74 m³/s) and a high degree of predictability. Although considered a mixed regime, only one real maximum occurs –

in June-July, when the highest water levels occur as a result of watershed melt water flow-off. Conversely, minimum discharges occur in winter, due to solid precipitation and nival retention (Figure 18).

2.2.2. Ribeira reach (Odelouca River, Portugal)

The Portuguese case study exemplifies the South-Western Europe flow regimes, with minimum flows in summer due to the seasonal lack of rain. This study site is located in the Odelouca River, near Ribeira village, with a studied length of close to 400 meters, at an altitude of about 132 meters. Riparian vegetation is typically Mediterranean, inhabited mostly by tamarisk (*Tamarix africana* Poir.), willow (*Salix salviifolia* Brot.) and narrow-leaved ash (*Fraxinus angustifolia* Vahl.). In the outermost floodplain areas it is also possible to find the emergence among riparian species of terrestrial species like cork oak (*Quercus suber* L.) or holm oak (*Quercus ilex* L. subsp. ballota). This case study features an intermittent river with a simple pluvial regime, where maximum mean monthly discharges occur in winter, while minimum discharges (commonly null) take place in summer. River flow is generally low, but discharge is highly responsive to rainfall and flash floods happen whenever there are heavy rain events (although mean discharge is 2.5 m^3/s, flash floods range between 80 and 480 m^3/s). This hydrological regime thus displays a great intra and inter-annual variability (Figure 18).

2.2.3. Terde reach (Mijares River, Spain)

Typical river flow regimes of mountain-fed catchments are illustrated by the Spanish case study, located in the Mijares River, between the villages of Sarrión and Mora de Rubielos. This site lies at an altitude of approximately 850 meters, where it presents a permanent river, 540 meters of which were surveyed. The floodplain vegetation is generally characterized by different willow species (*Salix eleagnos* Scop., *Salix purpurea* L. and *Salix alba* L.), black poplar (*Populus nigra* L.) and common reed [*Phragmites australis* (Cav.) Trin. ex Steud.]. Terrestrial species like juniper (*Juniperus spp.*), kermes oak (*Quercus coccifera* L.) or holm oak (*Quercus ilex* L. subsp. ballota) are also found within the one hundred-year flood area. This case study is characterized by a mixed pluvio-nival river regime with a low mean monthly discharge coefficient amplitude. This river flow regime displays two mean monthly discharge maximums,

one in January due to precipitation, and a more pronounced one in late spring originated by snowmelt (Figure 18).
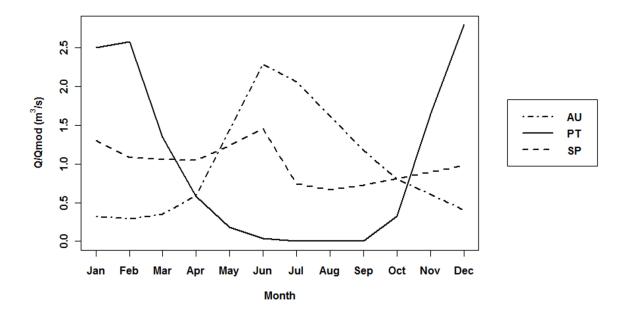
Figure 18. River flow regimes in the three considered study sites (Austria – AU, Portugal – PT and Spain – SP). Mean monthly discharges are presented as ratio Discharge (Q) / Mean annual discharge (Q_{av}) for 1960-1990 year period.

2.3. Climate change scenarios and expected hydrologic changes

In order to determine the deviation in riparian ecosystems caused by climate change, it is necessary to adopt a reference riparian patch mosaic from which to calculate riparian alterations linked to this stressor. To that end we considered a reference scenario, taking into account the popular and commonly used World Meteorological Organization (WMO) climate reference period of 1961-1990. This period is usually selected because it allows the comparison of future climate change regarding near present climatological conditions while having generally the best observational climate data coverage and availability from the periods considered meaningfully free from anthropogenic trends embedded (Mearns et al., 2001).

The climate change scenarios adopted in this study were based on the latest IPCC emission scenarios from which hydrologic modeling have been performed. As described in its Special Report on Emission Scenarios (SRES) (Nakicenovik and

Swart, 2000), this set of emission scenarios (A1 – *medium-high emission levels*, A2 – *high emission levels*, B1 – *low emissions* and B2 – *medium-low emissions*) attempts to reproduce the current knowledge in climate change science in order to characterize the range of probable driving forces and GHG emissions until 2100. Two of the above emission scenarios were selected for use as scenario templates in each case study, reflecting different intensities of climate change severity (Optimist and Pessimist scenarios) and spanning the existing uncertainties about future socioeconomic developments. In the light of the available data, the emissions scenario selection in each case study was determined in accordance with the Global and Regional Circulation Model scenarios whose results have been most consistent with the historical observations for each country, as regards temperature and rain forecasts in diverse climate change circumstances (see Santos et al., 2002; Moreno et al., 2005; Stanzel and Nachtnebel, 2010). Corresponding discharge anomalies in the study site flow regimes were then obtained from national climate change assessments in which hydrology was also envisaged. The anomalies were applied to the existing reference flow regime data for each study site by multiplicative factors obtained in those studies to obtain the corresponding study site scenario data series.

As a result, for the Kleblach reach study site, SRES B1 and SRES A2 emission scenarios were selected as Optimist and Pessimist respectively. The GCM model used as a basis for these scenarios was GCM ECHAM5 (Stanzel and Nachtnebel, 2010). The expected flow regime changes due to the projected meteorological alterations was determined by hydrological models based on information produced by the REMO-UBA regional climate model (Stanzel and Nachtnebel, 2010). The climate change scenarios for the Ribeira reach were grounded in the RCM HadRM3 results for the Optimist SRES B2 scenario and the Pessimist SRES A2 scenario, as presented for Portugal by Santos et al. (2002, 2006) (Santos et al., 2002; Santos and Miranda, 2006). The impact of climate change on freshwater assets was assessed using the Temez model – a simplification of the Stanford Watershed Model (Linsley and Crawford, 1960; Crawford and Linsley, 1966). Finally, for the Terde study site, the selected emission scenarios were also SRES B2 as the Optimist, and SRES A2 as the Pessimist. These were obtained from the Spanish modeling with the Hadley Centre Global Climate Model (HadCM3) as boundary conditions and regionalized with the PROMES regional climate model (Moreno et al., 2005). Hydrological scenarios were obtained from PATRICAL precipitation-runoff model results (Hernández, 2007). A summary of the hydrological

changes considered for the aforementioned climate change scenarios for each study site is presented in Table 4.

Table 4. Hydrological regime modifications accounted for the riparian vegetation modeling in the considered climate changes scenarios.

		Austria		Portugal		Spain	
		Change SRES A2 (Pessimist)	Change SRES B1 (Optimist)	Change SRES A2 (Pessimist)	Change SRES B2 (Optimist)	Change SRES A2 (Pessimist)	Change SRES B2 (Optimist)
Mean monthly discharge (%)	Winter (DJF)	38	26	-60	30	-30	-27
	Spring (MAM)	9	12	-80	-25	-24	-25
	Summer (JJA)	-17	-12	-80	-50	-32	-27
	Autumn (SON)	-11	-3	-80	-60	-33	-30
Minimum watertable elevation (m)		NE	NE	-4	-1	-0.27	-0.25
Flood duration		NE	NE	NE	NE	NE	NE

Values stand for deviation from Reference period (1960-1990). NE stands for non-expected changes.

2.4. Riparian vegetation modeling

For this task we used the state-of-the-art *Computer Aided Simulation Model for In-stream Flow and Riparian vegetation model*, commonly known as the *CASiMiR-vegetation* model (Benjankar et al., 2011). This tool is a dynamic rule-based spatially distributed model that supports its computation on fluvial disturbance in riparian vegetation – a concept that has been increasingly recognized since the late 1980's (Stanley et al., 2010) and whose influence is known to be a key cause of spatiotemporal variability in streams (White, 1979; Resh et al., 1988; Junk et al., 1989; Lake, 2000; Tockner et al., 2000; Stromberg, 2001). More precisely, this tool relates ecologically relevant hydrological elements (Poff et al., 1997) with riparian vegetation features that directly respond to chronic hydrologic alteration (Merritt et al., 2010), thus being able to reproduce local fluvial disturbance on an annual time step basis and determine the expected succession/retrogression phenomena in vegetation patches, depending on the fluvial physical driving forces to which they are subjected. The structure of *CASiMiR-vegetation* (Benjankar et al., 2011) consists of grid-based modules (Recruitment, Morphodynamic disturbance and Flood duration) functioning

with a Boolean logic framed by hard thresholds derived from expert judgment. Together, those modules mimic the succession/retrogression episodes experienced by patches when subjected to a particular fluvial disturbance stress.

A huge asset of this model is that modeling is performed by succession phase instead of site-specific features. This permits worldwide application (Benjankar et al., 2009; García-Arias et al., 2011; Politti et al., 2011; Egger et al., 2012; Rivaes et al., 2012) and eliminates divergences (e.g. species composition, ecoregion differences) that make generalized application unfeasible in many other models (see Merritt et al., 2010). Using this approach it is possible to obtain a homogeneous vegetation classification for the three case studies and thus permit a common appraisal of the modeling results. The adopted classification was first presented by García-Arias et al. (2013) (see this reference for a more detailed explanation of the vegetation types/succession phases transformation process), in which thirteen succession phases embedded on four succession stages and three succession series were acknowledged (Figure 19).
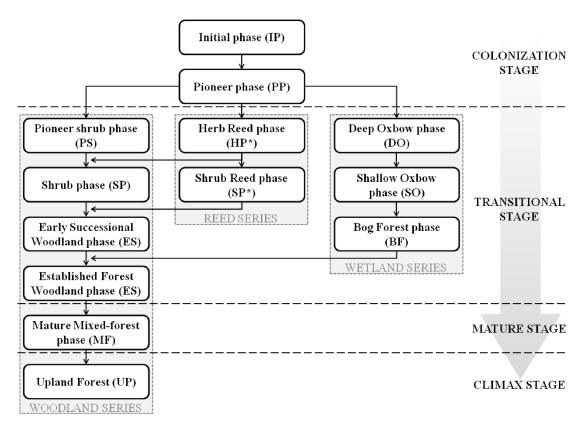
Figure 19. Common vegetation classification (by succession phase and stage) adopted for the three case studies, according to the existing vegetation series in each case study (adapted from García-Arias et al., 2013).

With this classification the model presented substantial positive results at the calibration/validation stage and also proved that a study site comparison analysis using standardized succession phases is possible. In addition, model uncertainty due to estimation errors in estimated parameter thresholds was determined not to be significant (Rivaes et al., 2013). *CASiMiR-vegetation* model calibration/validation for these cases is not presented here, as it is already thoroughly explained in previous studies (RIPFLOW, 2011; García-Arias et al., 2013).

The input data needed to run this tool are grid-based topography, maximum annual discharge shear stress, flood duration and mean/base water table elevation files. Our topography inputs were obtained by topographic surveys and were considered to be fixed during the modeling runs, so that riparian change evaluation could be endorsed solely to the hydrologic regime changes. Shear stresses and water table elevations in each study site were obtained by 2D hydraulic modeling, while flood duration was retrieved from daily recorded discharge data (RIPFLOW, 2011). Among the input data, shear stress stood out in terms of intra-scenario variability and was therefore analyzed for significant differences between scenarios. On the other hand, because minimum annual water table elevation and flood duration were considered unchanged within scenarios, we did not examine them by these means.

A simple method for appraising significant differences related to shear stress disturbance is to build confidence intervals for shear stress sample means in each scenario. We did this using two sample t-tests from the *R Stats package* in R environment (R Development Core Team, 2011).

Riparian vegetation modeling considered three modeling runs for each study site – namely the Reference, Optimist and Pessimist scenarios – starting from the same initial condition provided by the model. The expected 1990 riparian vegetation map was considered as the Reference scenario and was intended for use as a benchmark for assessing riparian deviations in the climate-change scenarios. The climate change scenarios (both Optimist and Pessimist) were characterized by the expected riparian vegetation maps at year 2100, under the corresponding climate-changed flow regimes. Once again, expected climate-changed riparian vegetation maps were obtained by modeling riparian vegetation under the likely river flow regimes in the 2071-2100 period. Riparian vegetation changes were analyzed by proportional change in total study site area and within each succession phase area, further denominated "specific

area cover anomaly", and referring to the difference between specific areas of succession phases in the Reference and climate-changed scenarios.

3. Results

For all study sites, the expected flow regime in each climate change scenario follows a pattern similar to that of its reference regimes (Figure 20). Having said this, some changes are noticeable and can lead to structural modifications in riparian woodlands. In the Austrian case, both scenarios forecast similar changes in the hydrological regimes. Winter and early spring mean discharges are likely to be higher than those in the reference period, whereas in the remaining months mean monthly discharge is expected to be lower. Nonetheless, water table elevations and flood durations are not expected to change significantly (Table 4). In the Portuguese case study, changes in the flow regime differ depending on the climate change scenario. This discharge variability is found in winter, when river flows are expected to be higher in the Optimist scenario, but lower in the Pessimist one. In the remaining seasons, both scenarios predict a reduced discharge compared to the corresponding Reference scenario, which in turn will contribute to a water table drop of about 1 and 4 meters in the Optimist and Pessimist scenarios, respectively. No flood duration changes are expected in this flow regime, as floods occur on a very short period of time (Table 4). Finally, in the Spanish case study both scenarios show a decreased discharge throughout the hydrological year, with very similar changes. Fluvial disturbance is attenuated, and reduced water availability will be experienced in the floodplains all year long. Water table elevations are expected to decline about 0.25m in the Optimist scenario and 0.27m in the Pessimist, while no changes were predicted concerning flood duration (Table 4).

Consistent with the expected climate change-induced flow regimes in each case study, maximum annual shear stress modifications in the study sites are also predicted. In fact, shear stress differences between scenarios proved significant with a 99% confidence level and corroborated earlier affirmations (Figure 21 and Appendix A – Table A1).
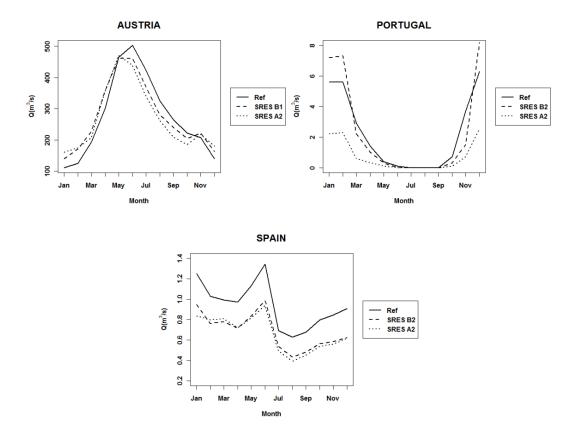
Figure 20. Reference and expected climate-changed hydrologic regimes in the considered study sites (discharge values stand for mean monthly discharges).
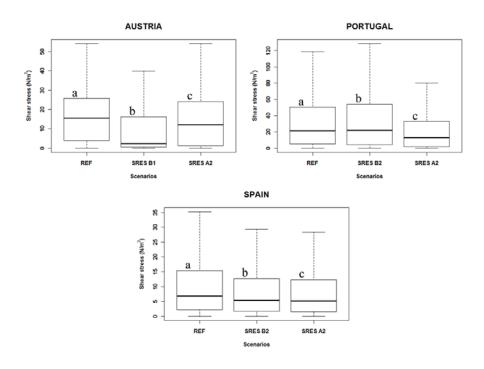
Figure 21. Scenarios of maximum annual discharge shear stress in each study site. Expected microhabitat shear stress of the maximum annual discharges in each study site according to the Reference, Optimist and Pessimist scenarios (whiskers stand for non-outlier extremes, box for 1st and 3rd quartiles, thick line for mean, and letters for significantly different groups).

Riparian vegetation modeling results show that, under the influence of climate-changed flow regimes, all the studied riparian ecosystems will experience structural changes in their riparian patch mosaics. Despite the fact that for the same modeling area (100-year flooded area), the three case studies achieved different stages in terms of vegetation development, the same tendency is perceptible in all of them. Novel succession phases are replaced by older and more hydric stress-tolerant ones in most cases; and wherever that replacement is not possible, riparian vegetation fades away, giving way to a complete retrogression to the Initial phase (Figure 22).
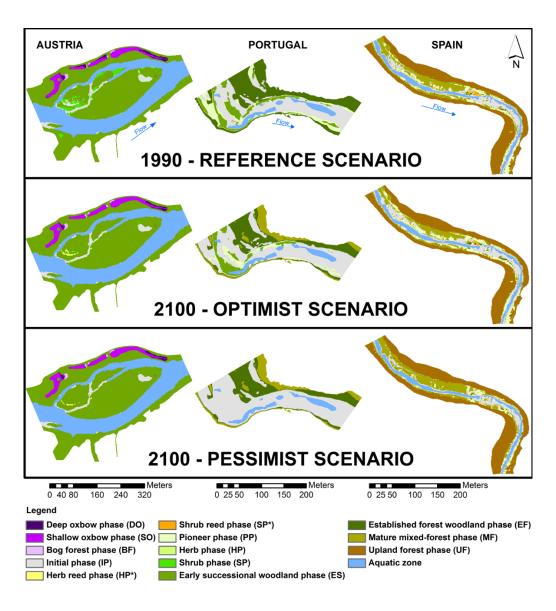
Figure 22. Riparian vegetation modeling results in each study site for the considered scenarios. Riparian vegetation modeling results in each study site for the Reference, Optimist and Pessimist scenarios.

Table 5 illustrates the proportional area covered by succession phases in each study site scenario. Austrian Reference scenario is characterized by the existence of three different vegetation series, mostly in a Transitional Stage (approximately 95%) and with little Colonization stage (near 5%). Riparian corridor is composed mainly of Woodland series (almost 87% of total area), the most common phase being Early Successional Woodland (ES) with about 82% of total area. Wetland series cover around 8% of total study site area, with Deep Oxbow phase (DO) with 1.5%, Shallow Oxbow phase (SO) with nearly 6%, and Bog Forest phase (BF) with 0.5%. The Initial phase (IP) represents almost 5% of total study site area. In opposition to the Reference scenario, slight changes are predicted in succession phases. As an example, in both Optimist and Pessimist scenarios the Woodland series Shrub Woodland Phase (SP) converts into Early Successional Woodland Phase (ES) with a consequent decline of approximately 4% in total area. In the case of the Wetland series, despite maintaining its cover area in all modeled scenarios, its succession phases adjust towards improved hydric stress adaptation. In fact, in both climate change scenarios the Deep Oxbow Phase (DO) decreases by 0.7% of total area, in favor of the Shallow Oxbow Phase (SO), which increases by the same amount in both scenarios. Reed series appear in the form of the Herb Reed phase (HP*), taking over areas once occupied by the Initial phase (IP) and where fluvial disturbance previously precluded vegetation establishment. In a climate change scenario, this succession phase achieves a habitat settlement ranging from 0.2% (in the Pessimist scenario) to 0.4% (in the Optimist scenario) of the total study site area.

The Portuguese case study presents a Reference scenario composed of Colonization and Transitional stages, each occupying approximately half the total area. Succession phases are present in different proportions, with Initial phase (IP) and Established forest phase (EF) occupying the majority of the study area (nearly 40% of total area each). In the considered climate change scenarios, the increase in the Colonization stage is proportional to climate change severity, due to the retrogression of younger phases, which attain growth of more than 60% in the Pessimist scenario. On the other hand, the Transitional stage deviation takes an inverse route, with a reduction to 33% in the Optimist scenario, and to less than 26% in the Pessimist one. Considering the specificity of the succession phase, it is noticeable that all succession phases are expected to experience moderate changes, ranging from around 3 to 24% of total area. Initial (IP) and Mature Mixed-forest (MF) phases swell in both scenarios, with the

former responsible for the biggest change in the riparian patch mosaic, specifically in the Pessimist scenario, where just this phase is responsible for a change in almost a quarter of the studied landscape. This increase occurs at the expense of the remaining phases, and even entails the total disappearance of the Pioneer (PP) and Early Succession Woodland (ES) phases in the worst scenario. The Established forest phase (EF) also reduces its cover area in the study site (roughly 13 and 12% in Optimist and Pessimist scenarios), but this time due to aging towards the Mature Mixed-forest phase (MF).

The Reference scenario in the Spanish study site is characterized by the existence of all the successional stages mentioned earlier and two succession series. Here, the Colonization, Transitional, Mature and Climax stages respectively comprise around 26, 3, 19 and 52% of the total area. With particular reference to the succession phases of the Woodland series, this case study is mainly represented by the Upland Terrestrial Forest (UF) and Mature Mixed-forest phases (together occupying nearly 70% of total area), while the remaining phases cover areas ranging 11% to 19% of the total area. Reed series cover almost 2% of the total area, namely in the form of a Shrub Reed Phase (SP*). In an Optimist climate change scenario, the Colonization stage increases by 5%, with a corresponding decrease in the Transitional and Mature Stages (0.4% drop-off in the former and nearly 5% in the latter). The Climax stage remains unaltered in this scenario. On the other hand, in the Pessimist scenario, Colonization and Transitional stages decline by approximately 7 and 1% of total area respectively, but the Mature and Climax Stages enlarge by approximately the same proportion of total area – namely 4% for the former and 4.5% for the latter. Where succession phases are concerned, minor changes are expected for the riparian patch mosaic in all the considered climate change scenarios, as none of the adjustments attain 5% of the total area. Major changes occur with the Pioneer (PP) and Mature Mixed-forest (MF) phases, but with no consistent trend. In fact, whereas in the Optimist scenario PP is expected to rise by nearly 3% and MF to decrease by about 5% of total area, in the Pessimist scenario PP faces a drop of almost 5%, but MF is reduced by more than 4% of total area. Changes in the Reed series represent a minute proportion of the total study area in both climate change scenarios.

Table 5. Changes in succession phase cover area according to the considered scenarios.

Succession series	Succession stage	Succession phase	Austria Reference scenario %	Austria Optimist scenario %	Δ	Austria Pessimist scenario %	Δ	Portugal Reference scenario %	Portugal Optimist scenario %	Δ	Portugal Pessimist scenario %	Δ	Spain Reference scenario %	Spain Optimist scenario %	Δ	Spain Pessimist scenario %	Δ
Any	Colonization stage	IP	4.9	4.1	-0.8	4.1	-0.8	37.3	46.6	9.3	61.3	24.0	14.0	16.0	2.0	11.3	-2.7
Any	Colonization stage	PP	-	-	-	-	-	13.1	7.5	-5.6	0.0	-13.1	11.7	14.8	3.1	7.1	-4.6
Woodland series	Transitional stage	HP	-	-	-	-	-	-	-	-	-	-	0.1	0.0	-0.1	0.1	0.0
Woodland series	Transitional stage	SP	4.3	0.5	-3.8	0.1	-4.2	-	-	-	-	-	-	-	-	-	-
Woodland series	Transitional stage	ES	82.4	86.6	4.2	87.2	4.8	11.6	8.3	-3.3	0.0	-11.6	0.3	0.2	-0.1	0.1	-0.2
Woodland series	Transitional stage	EF	-	-	-	-	-	38.0	24.8	-13.2	25.8	-12.2	0.9	1.4	0.5	0.7	-0.2
Woodland series	Mature stage	MP	-	-	-	-	-	0.0	12.8	12.8	12.9	12.9	19.0	14.2	-4.8	23.1	4.1
Woodland series	Climax stage	UF	-	-	-	-	-	-	-	-	-	-	52.3	52.3	0.0	56.8	4.5
Reed series	Transitional stage	HP*	0.0	0.5	0.5	0.2	0.2	-	-	-	-	-	0.0	0.0	0.0	0.1	0.1
Reed series	Transitional stage	SP*	-	-	-	-	-	-	-	-	-	-	1.7	1.0	-0.7	0.7	-1.0
Wetland series	Transitional stage	DO	1.5	0.8	-0.7	0.8	-0.7	-	-	-	-	-	-	-	-	-	-
Wetland series	Transitional stage	SO	6.4	7.1	0.7	7.1	0.7	-	-	-	-	-	-	-	-	-	-
Wetland series	Transitional stage	BF	0.5	0.5	0.0	0.5	0.0	-	-	-	-	-	-	-	-	-	-

Succession phase changes (area cover) in accordance with the considered case studies and scenarios. See Figure 5 for succession phase acronyms; percentage values relative to the total modeling area in each case study; Δ stands for scenario variation when compared to the Reference scenario.

However, although succession phase adjustments may not greatly change the riparian patch mosaic, standalone analysis does reveal profound alterations in the specific habitat area of each succession phase (Figure 23). This means that in the Austrian case some succession phases suffer extensive losses – e.g. the Shrub Woodland Phase (SP) experiences a specific decline in area of almost 90% in the Optimist scenario and faces near extinction (97.9% decline) in the Pessimist one. For the Wetland series, the Deep Oxbow Phase (DO) also faces a decrease in area of nearly 48% in both scenarios. In the Reed series there are noteworthy variations as well, but this time the Herb Reed phase (HP*) is expected to see a tenfold increase in area in the Optimist scenario and a fivefold one in the Pessimist scenario. In Portugal, at least a quarter of the areas of all the existing succession phases in the Reference scenario are expected to be modified in a climate change situation. The Pioneer (PP) and Early Successional Woodland (ES) phases are the most susceptible in this ecosystem, respectively suffering a specific area deprivation of approximately 43% and 28% in the Optimist scenario, while in the Pessimist scenario total retrogression may even occur. These decreases also lead to more than double the expansion of the Initial phase (IP) in the latter scenario. The Spanish case study is no exception to the other two, experiencing considerable succession phase changes. The area of Pioneer (PP) and Established Forest Woodland (EF) phases clearly increase, by almost 27 and 63% respectively in the Optimist scenario, while the Herb Woodland (HP) and Mature Mixed-forest (MS) phases are expected to suffer shrinkages in area of around 86 and 26% respectively. Succession phases in the Reed series are also prone to extensive reduction, with the Herb Reed (HP*) and Shrub Reed (SP*) phases losing roughly 89 and 41% of their specific areas. What is more, in the Pessimist scenario the Pioneer (PP) and Early Successional Woodland (ES) phases undergo a notable contraction in area of nearly 39 and 75% respectively. In this scenario the area of the Shrub Reed phase (SP*) is also likely to fall by approximately 58%, but it is estimated that the Herb Reed phase (HP*) will increase by almost 78%.
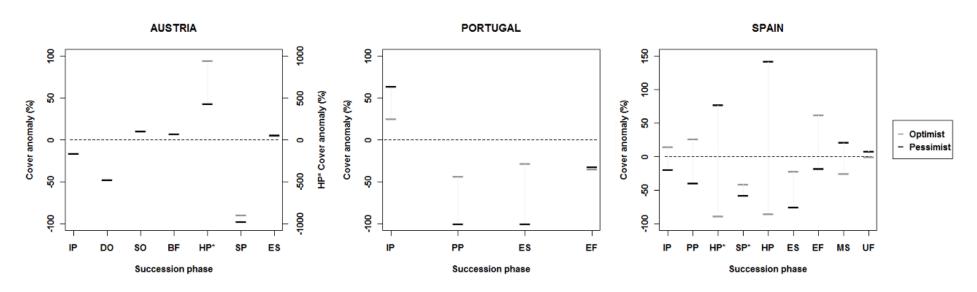
Figure 23. Specific area cover anomaly of succession phases. Specific area cover anomaly (%) of the succession phases in each study site and for the considered scenarios (see Figure 5 for succession phase acronyms).

4. Discussion

In all the considered cases there are expected changes in river flow regimes that can lead to significant effects on the hydraulic and hydrological conditions of riparian vegetation habitats, namely flood disturbance and hydric stress, which are effectively two of the most important conditioning factors in riparian dynamics (Tabacchi et al., 1998; Johnson, 1999; Bendix and Hupp, 2000; Stromberg et al., 2010a). River regimes powered mainly by snow melt or glacial thaw will experience minor increases in discharge. Winter discharges will be higher due to less nival retention, whereas summer discharges will fall due to the depletion of snow storage and the resulting decrease in melt water. In river regimes where rainwater is the main form of water alimentation, there is some uncertainty with regard to winter months, as not all rain forecasts agree (Alcamo et al., 2007b) and different flood disturbances are thus expected for this season, depending on the scenario. Nonetheless, both climate change scenarios expect riparian vegetation to be subjected to lower discharges and accentuated hydric stress in the remaining months of the year.

Accordingly, analyses of the microhabitat shear stresses of maximum discharges in each case study revealed significant differences between scenarios, proving that there will be a meaningful variation in flood morphodynamic disturbance in a climate change scenario.

The riparian vegetation modeling was performed using three different case studies contrasting in flow regime. Such flow regimes encompass the three main water alimentation forms of European rivers, according to Pardé's (Pardé, 1955) and L'vovich's (L'vovich, 1979) typologies, recently upheld by Wrzesiński (Wrzesiński, 2013). However, these case studies are representative of specific flow regime sub-types, which are not sufficient to make assumptions for the general trend of riparian vegetation changes driven by climate-changed flow regimes in Europe. Nevertheless, this study represents a first approach to portray that evolution.

To analyze the outcomes of the riparian vegetation model, one must regard a number of assumptions that first must be acknowledged. For this study, results should be understood within the context of vegetation patch dynamics, facing a certain scenario created by specific *CASiMiR-vegetation* model settings. The obtained forecasts need

to be interpreted more as an indicative trend rather than an exact prevision, due to the shortcomings of modeling such a high dynamic and complex system. The model was calibrated for each basin, considering that the vegetation patches evolution is essentially conditioned by the maximum discharge and by the minimum water table elevation registered in each year. To forecast that for different climate change scenarios the maximum annual discharge series in each basin were multiplied by a factor and the water table elevations were changed, according to the literature considered for the climate change scenarios (Santos et al., 2002; Moreno et al., 2005; Santos and Miranda, 2006; Hernández, 2007; Stanzel and Nachtnebel, 2010).

Despite the inherent stochasticity of fluvial systems, we opted by a deterministic modeling approach. Although, the non-consideration of the discharge sequence stochasticity of a flood event being a simplification, the maximum instantaneous discharge registered in each year seems to be the ultimate circumstance of morphodynamic forces driving the succession/retrogression dynamics of riparian woodlands (see Junk et al., 1989; Naiman and Décamps, 1997; Tabacchi et al., 1998; Bendix and Hupp, 2000; Friedman and Lee, 2002; Whited et al., 2007b; Stromberg et al., 2010a).

This restriction enhances the appreciation of broad features or general trends and allows the understanding on how specific components of the flow regime affect riparian vegetation (see Poff et al., 1997 for a better understanding). Consequently, based on this deterministic approach, we were able to eliminate the response variability caused by flow regime stochasticity and thus be able to address riparian responses to the discharges that are really important to condition the riparian vegetation dynamics.

Moreover, the fact that our modeling approach considers a fixed topographic input between years obviously represents a simplification of the multifaceted complex fluvial processes occurring within the riverbed. The flow patterns occurring over the banks of a river with riparian galleries and movable bed are very complex and difficult to model with accuracy. The models that consider the movable bed are still relatively inaccurate, due to the use of different empirical formulas, and to the difficulty in obtaining the representative granulometric curve of the different sediment patches and of the different sediment layers of the river bed, not to mention the possible occurrence of layers armoring. In the same line, the hydrodynamic patterns through the riparian

galleries are also very difficult to model, due to the vegetation heterogeneity and to the different bending resistance of vegetation species and of their succession phases to the flow velocity. The interaction between vegetation and sediment transport is, of course, still more complex. One example is the vegetation feedbacks, influencing the creation of fluvial landforms, trapping or stabilizing sediments, organic matter and the propagules of other plant species, i.e. acting as physical ecosystem engineers (Gurnell, 2014). Another effect particularly relevant in these case studies is the retrogression of transitional and mature stages, which are retrogressed mainly by side erosion and bank failure rather than mechanical disturbance. This is an aspect that will be very difficult to model and that was not considered in the present research.

In this context, the authors believe these complex effects should not be considered, so that the obtained results can reflect the influence of the main succession driving factors: maximum annual discharge and minimum water table elevation.

Besides, despite the recent recognition of those issues concerning the modeling of interactions between flow regime, vegetation and morphology (Gurnell et al., 2012; Camporeale et al., 2013), such processes were not yet implemented in the CASiMiR-vegetation model and would call for a specific research effort aiming at their integration in future model developments, not only within the climate change effects modeling but more generally within the riparian vegetation modeling context (Camporeale et al., 2013). But, the development of suitable models to simulate and analyze the biogeomorphologic feedbacks is still a priority in ecogeomorphology science agenda (Corenblit et al., 2011), as limited capacity remains to predict flow properties in vegetated channels, due to the great difficulty of linking complex dynamic vegetation structures to non-homogeneous hydrogeomorphic processes (Corenblit et al., 2007). Notwithstanding, in a similar study Politti *et al.* (Politti et al., 2014) suggested to consider a modeling period ranging from 5 to 25 simulated years, in order to work around those issues. According to this author, within this time frame the effect of the initial riparian landscape condition fades away after the 5th year while the non-consideration of the river morphological changes is not relevant before the 25th year.

Notwithstanding the previously stated, the performed vegetation modeling demonstrates that, for the considered flow regimes, contradictory changes are expected to occur in riparian ecosystems. While in snow-powered flow regimes

succession is most likely to occur right across the transversal gradient of the river, in rain-fed watersheds a more complex situation is expectable, with retrogression prevailing inside the channel and succession occurring in areas further from the river. In typical river flow regimes fed by mountain catchments, greater changes will likely occur in the older phases of the ecological succession, but, as other authors have pointed out (e.g., Auble and Scott, 1998; Johnson et al., 2000), results are not linearly correlated to any of the imposed stresses. In fact, lower flood disturbance and increased hydric stress do not result in a clear tendency in riparian vegetation structural amendment terms, thus showing that in this case shear stress and hydric stress don't explain successional dynamics by themselves.

Nor is the extent of change equal across the considered flow regimes. In both nivo-glacial and mountain-fed flow regimes, moderate changes in total area do occur, but some particular smaller variations in certain succession phases may not be enough to say whether this adjustment is due to model causal effects rather than model uncertainty or input errors. In fact, such a detailed analysis should be conducted carefully as the average model area balance error of succession phases in the three case studies was about 7% (RIPFLOW, 2011), especially in smaller and highly disturbed patches like younger succession phases. On the contrary, in Mediterranean pluvial flow regimes, succession area changes can be substantial and rivers with flow intermittency seem to be the most affected (Serrat-Capdevila et al., 2011), where succession phases can change *per se* almost a quarter of the total riparian patch mosaic.

Nonetheless, small changes in total area can mask dramatic habitat changes in succession phases within all the considered flow regimes. In fact, in the nivo-glacial regime-characteristic site, changes in succession phases can represent almost a tenth of the entire wetland areas, with large declines in some wetland succession phases, thus demonstrating that climate change will favor less water-dependent species. The same occurs in mountain-fed catchments, with succession phases experiencing specific area changes ranging from declines to near extinction, or to area boosts of about 50%. However, considering the variability of riparian responses to the climate-changed flow regimes in this case study, we are led to assume that in small river basins other factors may greatly influence riparian communities. These can include the availability of habitats provided by the river cross-section and the channel breath, or

even human-related pressures (Johnson, 1999; Aguiar and Ferreira, 2005; Ferreira et al., 2005).

All in all, climate-changed river flow regimes will most probably cause riparian vegetation amendments across rivers with similar flow regimes and even a general reduction in the areas covered by this vegetation. A common feature in all our case studies is that younger and more water-dependent phases are the most affected in a climate change scenario, whatever the forceful climate change or local environmental harshness may be. In snow-fed watersheds the main pathway for riparian vegetation appears to be succession, as minor summer floods cause less fluvial disturbance and greater hydric stress, which in turn allow vegetation to establish itself and develop to maturity, resulting in less water table-dependent phases. In pluvial flow regimes the tendency is consistently the opposite, despite some climatic uncertainties (Alcamo et al., 2007b). In this case, retrogression seems to be the main succession pathway for these communities, with large areas near river channels retrogressed to bare soil. Nevertheless, herein changes are not only due to the process of vegetation recycling to the Colonization stage, but also because of its aging to the Mature stage in the farthest floodplain areas. In mountain-fed catchments with mixed flow regimes the tendencies are not so clear and may reflect the existence of insufficient changes in flow regimes for there to be a clear change in their riparian communities. Meticulous analysis of the specific change in area in each succession phase showed that changes that may appear moderate when considering the total riparian patch mosaic can expose dramatic modifications when we look at the specific area changes in each succession phase. This means that many succession phases may face a serious threat in the future, when some of them will be confronted with complete annihilation. This outcome raises the question of maintaining viable populations of species that are important to conservation and are dependent on instream habitats. Additionally, more pronounced modifications – like the ones taking place in Southern European countries – are likely to occur in riparian communities that are dependent on pluvial flow regimes. These results are feasible expectations, inasmuch as similar riparian responses have been documented in vegetation assessments related to past flow regime events (Stromberg et al., 1996; Pettit et al., 2001; Stromberg, 2001; Shafroth et al., 2002; Dixon and Turner, 2006; Stromberg et al., 2007b; Stromberg et al., 2010a; Stromberg et al., 2010b). There are also existing forecasts that support our findings (Watson et

al., 1996; Frederick and Major, 1997; Primack, 2000; Moreno et al., 2005; Tague et al., 2009; Hoffman and Rohde, 2011; Schneider et al., 2011), although generally more superficially and with less detail regarding inner riparian community structure diversity. Climate change can therefore endanger specific riparian species, drive shifts in which exotics become dominant (Stromberg et al., 2007b; Hultine and Bush, 2011), or completely disrupt ecological succession in riparian ecosystems – something that can also lead to an increased risk to instream species survival (Broadmeadow and Nisbet, 2004) and flood hazards in downstream populations (Blackwell and Maltby, 2006).

These results also pave the way for improved knowledge about emerging topics that are as yet insufficiently studied in fluvial ecosystems (Winemiller et al., 2010). In this sense our results help substantiate the metacommunity Patch Dynamics Concept, which can be traced to Hutchinson's (Hutchinson, 1953) seminal ideas about non-equilibrium communities, and reinforces the notion that competitively inferior species are favored by patch disturbance, without which they would be replaced by competitively superior ones. It also helps understand the effects of patch dynamics across different river gradients, as well as the fact that species' life history attributes can influence community dynamics in response to disturbed flow regimes and changed habitat characteristics.

Finally, the results obtained by us through vegetation dynamics simulation can generate new questions stemming from riparian ecology concepts. The expected changes in the spatial ratio of different riparian types, with the likely suspension of succession in some cases, could lead to reflection on the interplay between the fluvial setting and vegetation (e.g., Muneepeerakul et al., 2007) – i.e. the relative dominance of non-equilibrium *versus* quasi-equilibrium processes (Bendix and Hupp, 2000). Our work also suggests new scientific questions regarding the potential feedbacks of novel habitats associated with an altered riparian vegetation mosaic, leading to changes in shear stress disturbance and hydrogeomorphic processes (Gran and Paola, 2001; Johnson, 2002), or in relation to potential alterations in the global functioning of the ecosystem and thus the services it provides.

Acknowledgments

SECTION III

SMALL-SCALE DRIVERS OF RIPARIAN

VEGETATION

CHAPTER 4

The role of river morphodynamic disturbance and groundwater hydrology as driving factors of riparian landscape patterns in Mediterranean rivers

Paper published in *Frontiers in Plant Science*

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Resumo

A perturbação fluvial, especialmente os fenómenos de cheia e de seca, são os principais condutores dos padrões sucessionais da vegetação ripária. Essas perturbações controlam a dinâmica da paisagem ripária através da direta interação entre o escoamento e a vegetação. O objetivo principal deste trabalho foi investigar os percursos específicos pelos quais a perturbação fluvial, classificada pelas suas componentes de hidrologia freática e perturbação morfodinâmica, conduz os padrões de paisagem ripária caracterizados pela localização (posição no corredor fluvial) e forma (forma física do polígono) dos polígonos de vegetação em rios mediterrânicos. Especificamente, este trabalho avalia como as diferentes componentes da perturbação fluvial afetam estas características em geral e particularmente em cada fase de sucessão da vegetação ripária. A hidrologia freática e a perturbação morfodinâmica foram definidas através de índices ponderados de duração e intensidade, calculados respetivamente, a partir dos níveis freáticos médios anuais e das tensões de cisalhamento dos caudais máximos instantâneos anuais, da última década. As interações entre as características da paisagem ripária e as perturbações fluviais foram avaliadas com análise fatorial confirmatória através de modelação de equações estruturais. Foram conceptualizados dois modelos hipotéticos para a localização e forma dos polígonos, que foram testados contra dados empíricos recolhidos a partir de 220 polígonos, em quatro locais de estudo diferentes. Ambos os modelos foram ajustados com sucesso, significando que conseguiram representar de forma adequada as relações entre variáveis. Para além disso, os modelos alcançaram um bom ajustamento aos dados observados, com base na avaliação de vários índices de qualidade de ajustamento. O modelo de localização dos polígonos explicou aproximadamente 80% da variabilidade das localizações, demostrando que a localização dos polígonos ripários é determinada principalmente pela hidrologia freática enquanto a perturbação morfodinâmica induz um efeito reduzido nesta característica. Numa análise multigrupos relativa às fases de sucessão da vegetação ripária, o modelo ajustado explicou mais de 68% da variabilidade dos dados, confirmando os resultados do modelo geral. O modelo de forma dos polígonos explicou aproximadamente 13% da variabilidade dos dados, revelando que as perturbações consideradas apresentaram menor influência na condução desta

característica. Não obstante, a hidrologia freática continua a ser o principal condutor da vegetação ripária de entre os dois fatores de perturbação considerados, apesar do aumento proporcional do efeito da perturbação morfodinâmica para aproximadamente um terço do efeito da hidrologia freática.

Palavras-chave: vegetação ripária, condutores ripários, perturbação fluvial, mediterrânico, análise fatorial confirmatória

Abstract

Fluvial disturbances, especially floods and droughts, are the main drivers of the successional patterns of riparian vegetation. Those disturbances control the riparian landscape dynamics through the direct interaction between flow and vegetation. The main aim of this work is to investigate the specific paths by which fluvial disturbances, distributed by its components of groundwater hydrology and morphodynamic disturbance, drive riparian landscape patterns as characterized by the location (position in the river corridor) and shape (physical form of the patch) of vegetation patches in Mediterranean rivers. Specifically, this work assesses how the different components of fluvial disturbances affect these features in general and particularly in each succession phase of riparian vegetation. Groundwater hydrology and morphodynamic disturbance were defined by time and intensity weighted indexes calculated, respectively, from the mean annual water table elevations and the annual maximum instantaneous discharge shear stresses of the previous decade. The interactions between riparian landscape features and fluvial disturbances were assessed by confirmatory factor analysis using structural equation modeling. Two hypothetical models for patch location and shape were conceptualized and tested against empirical data collected from 220 patches at four different study sites. Both models were successfully fitted, meaning that they adequately depicted the relationships between the variables. Furthermore, the models achieved a good adjustment for the observed data, based on the evaluation of several approximate fit indexes. The patch location model explained approximately 80% of the patch location variability, demonstrating that the location of the riparian patches is primarily driven by groundwater hydrology, while the morphodynamic disturbance had very little effect on this feature. In a multigroup analysis regarding the succession phases of riparian vegetation, the fitted model explained more than 68% of the variance of the data, confirming the results of the general model. The patch shape model explained nearly 13% of the patch shape variability, in which the disturbances came to have less influence on driving this feature. However, groundwater hydrology continues to be the primary driver of riparian vegetation between the two disturbance factors, despite the proportional increase of the morphodynamic disturbance effect to approximately a third of the groundwater hydrology effect.

Keywords: riparian vegetation, riparian drivers, fluvial disturbances, Mediterranean, confirmatory factor analysis

THE ROLE OF RIVER MORPHODYNAMIC DISTURBANCE AND GROUNDWATER HYDROLOGY AS DRIVING FACTORS OF RIPARIAN LANDSCAPE PATTERNS IN MEDITERRANEAN RIVERS

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Author Contributions

RR established the experimental design, conducted the experiment, analyzed data and wrote the manuscript. AP supervised the hydraulic modeling and reviewed the manuscript. GE supervised the ecological interpretation of the results and reviewed the manuscript. TF was the main supervisor and reviewed the manuscript.

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1. Introduction

Riparian ecosystems are dynamic systems found in flood-prone areas along rivers. They represent the transition between the aquatic and terrestrial ecosystems (Naiman and Décamps, 1997) and play a decisive role in riverine integrity (Van Looy et al., 2013). Riparian ecosystems rely greatly on the characteristics of the flow regime (e.g., Poff et al., 1997) and are notably susceptible to flow regime changes (e.g., Bejarano et al., 2012). The natural inter- and intra-annual variability of the flow regime determines the highly variable fluvial disturbances to which riparian vegetation respond structurally in the medium- to long-term (Whited et al., 2007a). Therefore, fluvial disturbances, i.e., the disruption imposed by the seasonal sequence of river flooding and drying (particularly their intensity and spatial extent), are the main drivers of the ecological succession of riparian vegetation (Corenblit et al., 2007). Accordingly, fluvial disturbances control the creation, development and recycling of vegetation patches (Bendix and Hupp, 2000). Fluvial disturbances are also important to maintain the ecological quality of riparian woodlands by providing services like reducing the occurrence of exotic plant species (Greet et al., 2015), promoting species diversity and richness in riparian bird communities (Merritt and Bateman, 2012) or even controlling the understory vegetation (Kamisako et al., 2007). Moreover, the dynamics of the disturbance pattern (Formann et al., 2013) substantiates the river processes that directly impact the riparian vegetation in its interactions with the surface and groundwater river flow (Camporeale and Ridolfi, 2006). The river stage is a proxy for fluvial disturbances. It fluctuates in the form of flood pulses (Junk et al., 1989) according to the hydraulic and hydrologic characteristics of the river. This effects the succession dynamics of riparian vegetation both physically and physiologically (Blom and Voesenek, 1996; Poff et al., 1997; Kozlowski, 2002; Džubáková et al., 2015) due to flood-induced stress through vegetation entrainment, uprooting, burial or anoxia (e.g., Friedman and Auble, 1999; Bendix and Hupp, 2000; Edmaier et al., 2011; Bendix and Stella, 2013). Also a consequence of the river stage is the oscillation of the groundwater level (Jansson et al., 2007). This determines a physiological effect by water stress control on plant growth and survival, affecting species differently according to their greater or lesser dependency on the connection of the root system with the groundwater table (Stromberg et al., 1996; Shafroth et al., 2000; Cooper et al., 2003; Baird et al., 2005; Lowry et al., 2011). The combination of all of these

morphodynamic and physiological factors determines a multiplicity of physical habitats that control the presence of riparian flow response guilds (Merritt et al., 2010; Sarr et al., 2011; Bejarano et al., 2012) in which discrete units of homogeneous vegetation occur in different succession phases. These succession phases are characterized by stands of specific ages, structural features and species compositions (Stanford et al., 2005). At a local scale, those are expected to be affected mainly by stream power and depth to groundwater (Bendix, 1999; Cooper et al., 1999; Bendix and Stella, 2013). Consequently, the riparian succession phase is a reliable indicator of the underlying hydraulic processes of fluvial disturbances, in which floods and droughts are the major stressors (Poff et al., 1997; Lytle and Poff, 2004; Stromberg and Boudell, 2013).

Currently, the function of riparian ecosystems and their interactions with their driving forces is well-understood (e.g., Gregory et al., 1991; Nilsson and Svedmark, 2002; Scott et al., 2005; Corenblit et al., 2007). However, the specific paths by which the drivers affect the riparian landscape have scarcely been investigated, especially regarding the local disturbances at a reach scale, or for Mediterranean flow regimes. In this context, this study aimed to investigate the effect of fluvial disturbances on two central elements of landscape ecology, the location (position in the river corridor) and shape (physical form of the patch) of riparian vegetation patches. Indeed, the patch location is important for the spatial characterization authenticity in patch-occupancy models (Fahrig, 2007) while the patch shape indicates the effect on many important ecological processes, such as colonization and growth (Hardt and Forman, 1989), landscape connectivity (Buechner, 1989), and most of all, ecosystem integrity associated with edge effects (e.g., Imre and Bogaert, 2004). As specific objectives, we were particularly interested in addressing the following questions. Can fluvial disturbances, particularly its components of morphodynamic disturbance and groundwater hydrology, affect the location and shape of riparian patches? How do these different components of fluvial disturbances affect these features? Is this effect on riparian vegetation similar in every succession phase? In order to address these questions we started by performing a thorough literature review to support the specification of our theoretical model constructs. After model specification, field surveys were carried out in three Mediterranean rivers to collect the necessary vegetation data. Finally, following data treatment, we analyzed the paths by which fluvial disturbances drive riparian vegetation patterns. By these means, we attempted

to provide essential knowledge on flow regime management for an enhanced riparian restoration, which is an indispensable and most promising way to restore natural processes in degraded rivers (Palmer et al., 2014).

2. Methods

The interactions between riparian spatial patterns and fluvial disturbances were analyzed by confirmatory factor analysis with structural equation modeling (SEM). SEM is a multivariate statistical modeling technique that combines factor analysis and regression analysis to validate fundamental theories with empirical data, and therefore provides a deeper analysis than traditional statistical methodologies (Malaeb et al., 2000). Furthermore, SEM has some interesting characteristics that overcome the standard first generation of multivariate statistical techniques, which are very useful for this kind of research. To begin with, SEM enables the incorporation of latent variables (also known as factors) in the analysis and tests the theoretical model constructs that they represent. Latent variables represent theoretical concepts that cannot be directly measured, such as ecosystem health or habitat suitability, but which are manifested by directly measurable variables (indicators or observed variables) that show the underlying variability of these theoretical concepts (Beaujean, 2014). SEM also enables the possibility of the simultaneous investigation of all of the effects and responses of the variables in the model construct, therefore providing a comprehensive picture of the system as a whole instead of the processes that comprise it. Finally, SEM takes measurement errors into account and hence offers better consistency and precision in parameter estimation (McCoach et al., 2007).

2.1. Model specification

The specification of the model consists of transforming the researcher's perceptions into the formal configuration of a structural equation model. This transformation is the most important and complex task in SEM because each of the following steps is grounded on the premise that the model designed is properly specified and that only a

correctly specified model can properly test the researcher's hypotheses (Ntoumanis and Myers, 2016).

According to the literature review presented in the Introduction, fluvial disturbance directly impacts riparian vegetation in two major ways, morphodynamic disturbance and physiological stress. A morphodynamic disturbance means surface flow-derived processes that cause physical vegetation damage, sediment burial or uprooting, and entrainment (e.g., Friedman and Auble, 1999; Edmaier et al., 2011; Bendix and Stella, 2013). Physiological stress is a consequence of the groundwater hydrology and impacts riparian vegetation as water stress caused by lowering of the level of groundwater table during a drought or by anoxia during flood periods (e.g., Stromberg et al., 1996; Shafroth et al., 2000; Cooper et al., 2003; Baird et al., 2005; Lowry et al., 2011). The current patch mosaic (i.e., the location and shape of the patches) of riparian vegetation is therefore the outcome of the existing conditions in the habitat, mainly as a consequence of the historical flow regime (Pettit et al., 2001). Consequently, we based ourselves on the literature review in the Introduction and our expert knowledge of riparian ecosystems to conceptualize two theoretical model constructs to address these research questions (Figure 24).

Morphodynamic disturbance (mrphd) and groundwater hydrology (grndh) are the exogenous variables that depict the flow regime and have a direct effect on the endogenous variables patch location (phslc; Figure 24A) and patch shape (phssh; Figure 24B). These variables cannot be measured directly, so they are considered latent variables expressed by manifest variables that are measurable. Because both morphodynamic disturbance and groundwater hydrology are controlled by the flow regime, they are expected to be correlated to some extent.

The physiological effects of the duration of flooding on vegetation were not expressly accounted for by the models because they were not expected to play an important role in the riparian ecological succession in the Mediterranean watersheds considered in this study. The characteristic flashiness of the pluvial flow regime defines very short flood durations in small catchment areas, and furthermore, are restricted to the winter (Tockner et al., 2009), which is the dormant vegetation season, when floods have less of an effect (Crawford, 2003).

Finally, the model constructs were hypothesized to be parsimonious as possible, and the latent variable indicators were reduced by the maximum amount. In fact, single or only a small number of latent variable indicators are normal in the natural sciences due to the nature of the data (Grace, 2006). This strategy was adopted not only to avoid identification problems (Schumacker and Lomax, 2010) but also to make use of only one or two of the best indicators as recommended and often sufficient (Hayduk and Littvay, 2012), as well as for the sake of the simplicity that must be sought for any ecological model (Jackson et al., 2000).
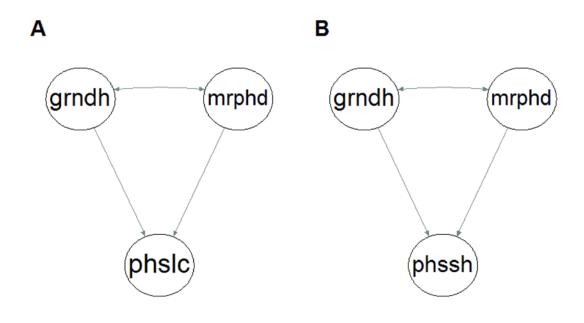
Figure 24. Conceptual model construct of riparian patch location (A) and shape (B). Ellipses represent the following latent variables: groundwater hydrology (grndh), morphodynamic disturbance (mrphd), patch location (phslc) and patch shape (phssh). Single-headed arrows stand for direct relationships and double-headed arrows between variables for existing unexplained correlations.

2.2. Model identification

The objective of model identification is to determine if a theoretical model construct enables the unique estimation of the requisite parameters from existing non-redundant information in the data. Consequently, to assure the identification of the proposed models, the factor loadings of the latent variables with only one indicator were set to 1

and the corresponding indicator error variance was set to 0 (Beaujean, 2014). Hence, 4 factor loadings exist for the specified models, 4 measurement error variances, 2 path coefficients, 1 correlation between the latent variables and 3 equation error variances – a total of 14 parameters that must be estimated. Each of the model constructs has only 8 free parameters (relationship coefficients to be estimated from the collected data) and 4 observed variables, implying that there are $(8 \times 4)/2 = 16$ pieces of non-redundant information. Because only 14 parameters must be estimated, the models are considered overidentified and therefore the model identification is verified. Additionally, to have sufficient variability to estimate the model parameters, the sample size (N) should follow the rule of at least 20 cases for each free parameter that must be estimated (Jackson, 2003). Accordingly, in this situation N should be at least 160 observations. Notwithstanding, the usual minimum sample size for SEM studies is approximately 200 cases (Kline, 2011).

2.3. Data collection

Four study sites were selected in natural conditions of riparian vegetation and flow regime (Figure 25). Upstream of the study sites, the main land uses in the watersheds are planted forests and natural shrublands, with very sparse villages and no noteworthy industry. In all cases, the flow regime was considered unregulated and typically Mediterranean, with a low winter flow interspersed by flash floods, and a very low and often intermittent summer flow (Bonada and Resh, 2013). The woody riparian species composition was similar in the study sites, comprising mainly willows (*Salix spp.*) and ashes (*Fraxinus angustifolia*). Notwithstanding, the four sites encompassed diverse fluvial geomorphologies, watershed features and river sizes (Table 6).
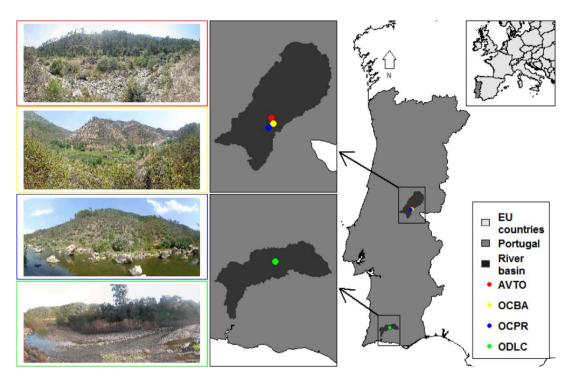
Figure 25. Location and characterization of the study sites AVTO (red), OCBA (yellow), OCPR (blue) and ODLC (green).

A complete survey of the topography and riparian vegetation was performed at each study site. The surveys were done on river reaches 300 to 500 meters long (depending on the river width maintaining a ratio between length and main channel width of 10 to 20) and extended laterally to the area flooded by a 100-year recurrence interval (normally determining study site widths of approximately 70 to 110 meters). This area was determined by modeling such a flood in each study site using the hydrodynamic model River2D. The topography was surveyed with an elevation detail of 20 cm using both total stations (Leica TPS400) and Global Navigation Satellite Systems (Leica 500 GPS, composed of two double-frequency-to-real-time SR 530 RTK antennas L1 and L2 AT 502). Riparian vegetation surveys were performed using a sub-meter handheld GPS (Ashtech, Mobile Mapper 100) to outline and georeference all the existing vegetation patches. Vegetation inventories in each patch sought to characterize succession phases by vegetation attributes and included phanerophyte species identification (Table 6). Then, each vegetation patch was classified by its succession phase, based on the vegetation type and the patch age. The succession phase classification methodology followed Rivaes et al. (2013) in which vegetation types were defined by indicator species and patch age by dendrochronological methods. A total of

220 vegetation patches in the aquatic, bank and floodplain zones were assessed. Four succession phases were found at all the study sites, namely, initial phase (IP), pioneer phase (PP), early succession woodland phase (ES) and established forest phase (EF). Altogether, those succession phases account for the existing ecological succession phases of riparian woodlands in those Mediterranean rivers (García-Arias et al., 2013). In detail, IP is characterized by areas with less than 50% vegetation cover and the absence of woody species. PP is typified by the recruitment areas of woody species and ES is characterized by the presence of well-established softwood pioneer individuals such as willows. EF is found in patches presenting hardwood species such as ashes, along with a well-defined understory stratum (Table 6).

The hydraulic parameters were obtained with the River2D model (Steffler et al., 2002) and were used to create the shear stress maps of the annual maximum instantaneous discharges and the mean annual water table elevations. The River2D is a hydrodynamic 2D model based on the depth averaged Saint Venant equations that computes the depth and the discharge intensities in the x-y directions. This tool was developed for application in natural rivers and features wet-dry area solution capabilities by combining surface flow and groundwater flow equations to compute the free elevation above and below the ground. River2D also incorporates a bed resistance model and a transverse shear model. In the former model, bed shear stresses are assumed to be related through the effective roughness height to the magnitude and direction of the depth-averaged velocity. The advantage of using the roughness height as the resistance parameter is that it remains constant over a wide range of depth. In the transverse shear model, the depth-averaged transverse turbulent shear stresses are modeled using a Boussinesq type eddy viscosity formulation. A complete description of the model is provided by Ghanem et al. (1996). Shear stress has been widely used as a fundamental proxy for soil erodibility, morphodynamic disturbance and drag imposed on vegetation by river flows. In natural channels, shear stress is considered to be balanced by three resistance components, namely, viscous drag on the ground surface on particles, pressure drag associated with large non-vegetal boundary roughness and drag on vegetal elements (Temple et al., 1987). The maps produced by River2D had a precision of a quarter of a square meter and were used to compute the morphodynamic disturbance index (MDi) and the groundwater depth index (GWDi) developed by Egger et al. (2014, 2016). Both are time and intensity

weighted indexes (TIWI) that characterize the processes of groundwater hydrology and morphodynamic disturbance of the historical flow regime in the past decade at each study site. These indexes are proxies for the long-term ecosystem processes of physiological stress and geomorphic-mechanical disturbance that effect riparian communities and provide a parameter that allows a dynamic analysis of riparian ecosystem patterns. Mean values of MDi and GWDi were calculated for the area covered by each of the vegetation patches recorded. The patch features were obtained using the *ArcGis 9.2* (Environmental Systems Research Institute, 2010) and its *Patch Analyst* extension (Rempel et al., 2012). The relative positioning of the patches in the study sites were characterized by height (THAH) and distance (THAD) to thalweg. The patch shape metrics selected were the patch perimeter (PERIMTR) and the mean patch fractal dimension (MPFD). PERIMTR is an edge metric that provides a measurement of the dimensions and amount of edge created by each patch. This variable significantly affects many ecological phenomena and the analysis of spatial patterns in landscape ecological research (McGarigal and Marks, 1994). The MPFD is a shape metric that represents the geometric complexity of the patches and provides information regarding the formation and quality of the patches (Imre and Bogaert, 2004). Finally, the patch data were compiled and a data matrix was built and uploaded in the R environment (R Development Core Team, 2011) for subsequent data validation and treatment.

Table 6. Characterization of the study sites.

		OCBA	OCPR	AVTO	ODLC
Watershed area (km^2)		779	1037	177	186
Distance do source (km)		63	66	30	34
Mean discharge (m3/s)		7.9	11.0	2.3	3.2
Mean annual maximum instantaneous discharge (m^3/s)		322	457	95	116
Mean slope (m/m)		0.003	0.006	0.012	0.002
Main substrate type		BOULDERS	LARGE BOULDERS	BOULDERS	COBBLES
Total succession phase area (%)	IP	37.2	30.5	26.37	25.8
	PP	23.2	8.0	6.67	5.7
	ES	11.3	22.1	16.2	18.6
	EF	28.2	39.4	50.8	49.9

Species inventory (mean coverage %)*

	IP	PP	ES	EF	IP	PP	ES	EF	IP	PP	ES	EF	IP	PP	ES	EF
Alnus glutinosa	-	0.1	1.8	-	-	0.1	-	-	-	2.1	10.4	-	-	-	-	-
Cistus spp.	-	-	-	-	-	-	-	1.2	-	-	-	1.2	0.1	-	-	0.2
Crataegus monogyna	-	-	-	7.6	0.3	-	-	0.2	1.3	1.4	1.1	2.0	-	-	0.3	2.3
Erica spp.	-	2.4	4.0	8.8	0.3	-	-	16.6	0.1	-	1.4	14.0	-	-	-	-
Ficus carica	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1.2
Flueggea tinctoria	0.5	10.6	21.5	1.9	1.8	4.6	21.1	-	0.3	28.1	26.8	-	-	-	-	-
Fraxinus angustifolia	-	0.2	5.1	27.4	0.3	-	0.5	24.6	0.3		3.2	1.6	1.4	4.0	9.0	45.8
Olea europeae	-	-	-	-	-	-	-	-	-	-	-	1.0	0.2	-	0.1	10.6
Phillyrea spp.	-	-	-	2.0	0.5	-	-	1.2	-	-	-	5.2	-	-	-	0.2
Pinus pinaster	-	-	-	0.1	-	-	-	0.4	-	-	-	1.2	-	-	-	-
Quercus rotundifolia	-	-	-	-	-	-	-	0.2	-	-	-	1.0	-	-	-	3.5
Quercus suber	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	3.0
Rosa canina	-	0.6	1.0	0.8	-	-	-	1.0	1.0	-	0.4	1.0	-	0.1	0.8	1.6
Salix spp.	0.5	10.0	41.3	-	1.8	18.0	47.9	-	-	19.6	35.0	0.4	1.4	0.1	27.7	-
Tamarix africana	-	-	-	-	-	-	-	-	-	-	-	-	2.9	10.6	22.8	14.5

* Species inventory only include phanerophyte species with at least 1% coverage in one of the succession phases of the four study sites. All the species mentioned are native.

2.4. Data validation and treatment

The variables were log transformed to remove bias and to bypass model under-identifiability imposed by disproportionate scales (McDonald and Ho, 2002). Outliers for which no justification was found were removed. Data were subsequently checked for compliance for the statistical assumptions for the SEM. Data normality was assessed with the Shapiro-Wilk normality test (Shapiro and Wilk, 1965), augmented with an assessment of data skewness (sk) and kurtosis (ku). Sk and ku were also used to assess the data normality to some extent, providing information about the distribution of the variables, i.e., the symmetry and the "peakedness" of the distribution. In this particular case, even after the data were transformed, some of the variables considered continued to have normality issues and did not pass the Shapiro-Wilk normality test. However, the magnitude of the skewness and kurtosis of the variables were always below 0.88 and 1.01, respectively. No consensus exists as yet about the maximum magnitude of sk and ku that does not undermine the reliability of the conclusions about the model fit and the parameter estimates (Finney and DiStefano, 2006) but the most conservative thresholds found in the literature are maximum absolute values of 1 and 1.5, respectively (e.g., Schumacker and Lomax, 2010; Kline, 2011). Furthermore, for large sample sizes (N≥40) non-normality is not considered problematic (Pallant, 2002) and can be ignored (Altman and Bland, 1995). Consequently, the data were considered to be normal. The linearity between the variables was assessed using the Harvey-Collier and Rainbow tests for linearity, as well as Ramsey's RESET test for functional form. The majority of the variable relationships passed all three tests, but only two relationships were linear in at least one of the tests with a 99% confidence level. Consequently, the variables were considered to be linearly related. Independence was tested using Moran's autocorrelation coefficient (Moran, 1950) and the Mantel test (Mantel, 1967) for spatial correlation. The data were considered independent, because both Moran's coefficient and the Mantel test did not demonstrate any spatial correlation of the riparian succession phases between sites or at each site, with a confidence level of 99%. Homoscedasticity was assessed by a plot analysis of the residuals versus the predicted values. No substantial problems were found by a visual assessment of the plots and therefore, the data was considered homoscedastic. Multicollinearity was evaluated by calculating the Variance Inflation Factor (VIF) for each variable. The VIF

is a widely used parameter to determine the degree of multicollinearity between sets of observations (Liao and Valliant, 2012; García et al., 2015; Salmerón Gómez et al., 2016), and provides an indication of the effects of multicollinearity on the variance of the regression coefficients. A threshold of 10 is the most common value used as an indication of multicollinearity and a lower VIF is indicative of inconsequential multicollinearity (O'Brien, 2007). Multicollinearity was not considered problematic since the VIF values ranged from 1.38 to 6.11.

2.5. *Model estimation and evaluation*

Model estimation was performed using the *lavaan* package (Rosseel, 2012) running in the *R* environment (R Development Core Team, 2011). The models were fitted with the maximum likelihood estimator, the most widely used in SEM. This estimator is unaffected by data transformation and has asymptotic properties, hence the minimum variance, unbiasedness, efficiency and consistency (Myung, 2003). The maximum likelihood estimator assumes multivariate normality but is appropriated and robust even if non-normality exists (Yuan and Bentler, 2005) and still produces centered estimates of the parameters (Finney and DiStefano, 2006) unless sk and ku are too severe (Kline, 2011), which was not the case for our data.

The model fit was assessed using the goodness-of-fit chi-square (χ^2) statistic at a significance level α=0.05. This model test statistic tests the null hypothesis that the population covariance matrix does not differ significantly from the model-implied covariance matrix. A failure to reject the null hypothesis means that the model is consistent with the sample data matrix, thus supporting the model assumptions. χ^2 is the most commonly used statistic for the model fit in SEM to appraise the overall model fit (Hu and Bentler, 1999) and it works particularly well for sample sizes between 75 to 200 cases (Kenny, 2015). It tends to be significant for larger sample sizes due to its sensitivity to sample size (Schumacker and Lomax, 2010). Additionally, the correlation residuals of the model fit were established to confirm the explanatory power of the model for the specific associations observed. As a rule of thumb, a correlation residual with an absolute value greater than 0.10 is a sign of a poor explanation of the corresponding sample correlation (Schumacker and Lomax, 2010). In addition to χ^2, several other approximate fit indexes were considered to provide further insights about

the model fit such as the Steiger-Lind Root Mean Square Error of Approximation (RMSEA; Steiger, 1990), the Jöreskog-Sörbom Goodness of Fit Index (GFI; Jöreskog and Sörbom, 1982), the Bentler Comparative Fit Index (CFI, Bentler, 1990), the Tucker-Lewis Index (TLI; Tucker and Lewis, 1973), the Standardized Root Mean Square Residual (SRMR) and the ratio χ^2/df. Taken together, these are among the most common approximate fix indexes that exist in the SEM literature (McDonald and Ho, 2002).

A multigroup analysis was also performed to determine how the considered disturbance factors specifically affect each riparian succession phase. This analysis was preceded by an evaluation of model measurement invariance, to judge the validity of the proposed comparisons, along with the computation of CFI variation (ΔCFI) to confirm the evaluation. During the multigroup analysis, the proposed models were individually fitted to each succession phase because the sample size of each group was considerably lower than the recommended sample size for these analyses. Consequently, a 95% confidence interval for the mean value of the group-specific model parameters was built in these cases based on larger sample sizes generated by the bootstrapping resampling method with 1000 bootstrap draws. All of the relationships between variables were analyzed using standardized coefficients for a better understanding of the direct effects between the variables (Grace and Bollen, 2005).

3. Results

3.1. Patch location

Figure 26 shows the proposed model construct for patch location, which was successfully fitted with a non-significant χ^2 test (p-value=0.221). Accordingly, the population covariance matrix was not significantly different from the covariance matrix estimated by the model at a confidence level of 95%. The additional approximate fit indexes also supported this outcome and provided a perception of the goodness of the adjustment of the proposed model for the sampled data. The CFI, TLI, GFI and RMSEA indicated a very good adjustment, but the SRMR and χ^2/df indicated that this result

was a good adjustment (see Schumacker and Lomax, 2010; Kline, 2011; Marôco, 2014 for summaries on the fit classification indexes). Furthermore, all of the correlation residuals were less than 0.10, which suggested that the model could properly explain the corresponding sample correlations.

In this model, morphodynamic disturbance is manifested on MDi and groundwater hydrology on GWDi. The latent variable of patch location is manifested on patch distance to thalweg (THAD) and height above thalweg (THAH), which indicates the patch positioning along the river stretches. The fitted model explains approximately 80% of the patch location variability, which implies that these variables contribute a considerable amount of information regarding patch location. The results show that the location of the riparian patches is primarily driven by groundwater hydrology (standardized path coefficient=0.930), whereas the morphodynamic disturbance seems of very little importance (standardized path coefficient=0.061) to patch location. Regardless, groundwater hydrology and morphodynamic disturbance are moderately correlated (-0.592). Consequently, riparian patches located at larger horizontal and vertical distances from the river thalweg appear to be subject to greater groundwater hydrology disturbances. Furthermore, these results show that changes in groundwater hydrology may lead to shifts in the location of succession phases, the eventual loss of the more vulnerable ones and the following extinction of its characteristic species.
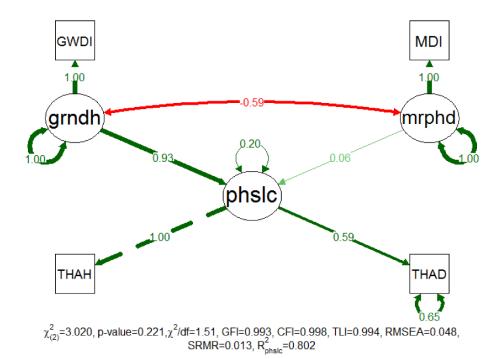
Figure 26. Completely standardized solution of the fitted model for patch location. Standardized path coefficients are shown in red or green accordingly to the sign of the relationship. Color intensity and arrow thickness are proportional to relationship magnitude. Continuous and discontinuous arrows stand for free and fixed-unit path coefficients. GWDI is an abbreviation for groundwater depth index, MDI for morphodynamic disturbance index, grndh for groundwater hydrology, mrphd for morphodynamic disturbance, phslc for phase location, THAH for height above thalweg, and THAD for distance to thalweg.

The results for measurement invariance show that the patch location model indicates a weak invariance between study sites (H0: the model is invariant between study sites, not rejected for weak invariance with a p-value=0.3345). This finding means that the model construct is similar at all of the study sites and that for a given indicator, the factor loadings are significantly the same between the study sites. The ΔCFI (0.001) for the loadings is less than 0.01, indicating once more that invariance should not be rejected (Cheung and Rensvold, 2002). The model shows strong measurement invariance for the succession phases (the p-values for the weak and strong invariance tests are 0.4428 and 0.1664, respectively). The ΔCFI for the loadings and intercepts is correspondingly, 0.001 and 0.004, again smaller than the proposed threshold of 0.01 and therefore support the results of the invariance tests. A weak invariance allows the comparison of the relationships between the latent variables across groups while a strong invariance allows for the inter-group comparison of latent variable means and covariances (Chen, 2008).

The proposed model was successfully fitted in the multigroup analysis (Figure 27). The χ^2 test was not significant (p-value=0.165) and the approximate fit indexes indicated the good adjustment of the model. The model explained more than 68% of the data variance, for group sample sizes of 74, 52, 72 and 22, respectively, for IP, PP, ES and EF. As in the general location model, for every succession phase, both disturbances are still well correlated and the groundwater hydrology is the main driver of patch location (the standardized path coefficients were greater than 0.81 for all succession phases). In contrast, the morphodynamic disturbance had a residual effect on the succession phases, except for ES (standardized path coefficient=0.207), for which an increase in the morphodynamic disturbances determined greater distances to thalweg. This means that morphodynamic disturbance pushes away this succession phase to a more distant location, thus preventing vegetation encroachment. In detail, IP is characterized by recently disturbed patches where woody vegetation is starting to establish. The seedlings survive according to the recruitment box (Mahoney and Rood, 1998) and therefore, the link with the water table elevation is tight. If the recruitment survives, the patches evolve to PP after approximately two years. Individuals have now a settled root system but still vegetate inside the active channel, where the groundwater remains within reach almost all year round, accounting for the less pronounced link with the groundwater hydrology. The ES patches live in a survival limbo between the groundwater hydrology and morphodynamic disturbances. The indicator species of this succession phase (*Salix spp.*) are obligate phreatophytes well adapted to flow disturbances (González et al., 2010), whose existence is compelled by morphodynamic disturbances to the limit of the tolerated distance from the groundwater table. Consequently, these patches are considerably more likely to be negatively affected by groundwater level fluctuations and are therefore highly dependent on groundwater hydrology. In turn, the typical facultative phreatophytes that characterize EF confer to this succession phase less dependency on this factor and a survival advantage over previous succession phases.
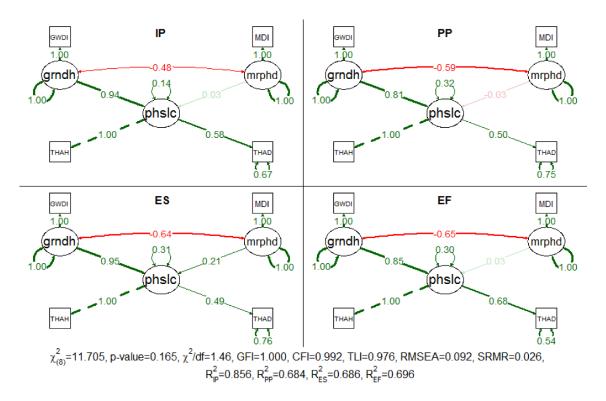
Figure 27. Completely standardized solution for the fitted model for patch location in each succession phase (IP – Initial phase, PP – Pioneer phase, ES – Early succession phase and EF – Established forest phase). Standardized path coefficients are shown in red or green accordingly to the sign of the relationship. Color intensity and arrow thickness are proportional to relationship magnitude. Continuous and discontinuous arrows stand for free and fixed-unit path coefficients. GWDI is an abbreviation for groundwater depth index, MDI for morphodynamic disturbance index, grndh for groundwater hydrology, mrphd for morphodynamic disturbance, phslc for phase location, THAH for height above thalweg, and THAD for distance to thalweg.

The 95% confidence intervals of the bootstrapped coefficients confirm the discrepancy in the effect between the groundwater hydrology and morphodynamic disturbances on the location of the succession phase patches (Table 7). In fact, the magnitude of the effects of the two disturbances is completely distinct in the first three succession phases and always higher for groundwater hydrology in all succession phases. Furthermore, one cannot rule out the possibility of that a morphodynamic disturbance would have no effect on the location of the succession phases, unlike groundwater hydrology, which can be the only driving factor of the location of the succession phases as indicated by the proposed model. Succession phases have different heights and distances to thalweg, so it is possible to infer the location of the succession phases along the river stretch despite some degree of overlap. IP is the closest succession phase to the thalweg (the standardized values for mean distance and mean height to

thalweg is 3.220 and 3.406, respectively), followed by ES (4.669 and 4.731), PP (4.787 and 5.984) and EF in the outer parts of the river (5.499 and 6.824).

Table 7. 95% confidence intervals of the bootstrapped coefficients for the multigroup patch location model.

	Initial phase	Pioneer phase	Early Successional Woodland phase	Established Forest phase
Patchloc~Groundhydro	0.848 ; 1.025	0.531 ; 1.364	0.615 ; 1.080	0.267 ; 1.149
Patchloc~Morphodist	-0.130 ; 0.191	-0.406 ; 0.503	-0.005 ; 0.413	-0.379 ; 0.398
THAD	2.568 ; 4.242	3.831 ; 6.475	3.770 ; 5.903	5.437 ; 5.561
THAH	2.861 ; 4.216	5.393 ; 6.029	3.556 ; 6.253	6.743 ; 6.905

3.2. Patch shape model

The patch shape model was successfully fitted with a non-significant χ^2 test (p-value=0.078), and the additional approximate fit indexes indicated a good adjustment classification (Figure 28). GFI, CFI, TLI and SRMR indicated that this adjustment was very good while χ^2/df and RMSEA classified it as acceptable. The latent variables of fluvial disturbance are manifested as GWDi and MDi while the latent variable patch shape is manifested as patch perimeter (PERIMTR) and the mean patch fractal dimension (MPFD). Nevertheless, this model only explained approximately 13% of the patch shape variability and the disturbances accounted for have smaller effect on this feature (the sum of standardized patch coefficients is 0.562). The groundwater hydrology continues to be the main driver of the two disturbance factors, despite that the effect of the morphodynamic disturbances (standardized path coefficient=-0.139) has now proportionally increased to approximately a third of the effect of the groundwater hydrology (standardized path coefficient=-0.423).
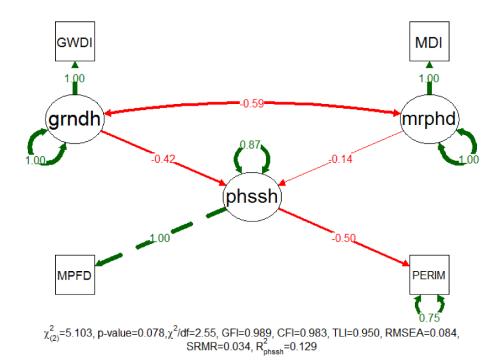
Figure 28. Completely standardized solution of the fitted model for patch shape. Standardized path coefficients are shown in red or green accordingly to the sign of the relationship. Color intensity and arrow thickness are proportional to relationship magnitude. Continuous and discontinuous arrows stand for free and fixed-unit path coefficients. GWDI is an abbreviation for groundwater depth index, MDI for morphodynamic disturbance index, grndh for groundwater hydrology, mrphd for morphodynamic disturbance, phssh for phase shape, MPFD for mean patch fractal dimension, and PERIM for patch perimeter.

The investigation of the measurement invariance revealed a weak and strong invariance between study sites (p-values of 0.268 and 0.844), respectively. Likewise, the ΔCFI for loadings (0.001) and intercepts (0.001) were smaller than the proposed threshold of 0.01, providing confirmation of both levels of measurement invariance. The model only shows weak measurement invariance (p-value=0.3828) for the succession phases supported by a ΔCFI for the loadings of 0.0004.

Notwithstanding, the patch shape model did not achieve a successful fit in the multigroup analysis. The χ^2 test was significant ($\chi^2_{(8)}$=20.791 and p-value=0.008), meaning that the population covariance matrix was significantly different from the model-estimated covariance matrix. Also the approximate fit indexes indicated adjustment problems with contradictory classifications, specifically, an unacceptable fit by the TLI (0.757) and RMSEA (0.171), an acceptable fit by χ^2/df (2.60), CFI (0.919) and SRMR (0.056), and a perfect fit by GFI (1.000). In addition, the explanatory power

of the model was very low for all of the succession phases (0.066 for IP, 0.048 for PP, 0.034 for ES and 0.225 for EF). Taken together, these results demonstrated that the proposed model did not adequately explain the patch shape dynamics of the riparian succession phases in the Mediterranean rivers.

4. Discussion

The current study focused on understanding the drivers of landscape features of Mediterranean riparian vegetation on a local scale, specifically, the location and shape of the patches. The approach adopted relied on testing the existing theories about the local drivers of riparian vegetation and its succession phases. Accordingly, the model-implied covariance matrices of the hypothetical model constructs were tested with the population covariance matrices built from the vegetation data collected at four different Mediterranean case studies. Both models were successfully fitted, confirming that the structural equation models were consistent with the sample data and were therefore able to correctly characterize the riparian patch dynamics of the considered Mediterranean rivers. Nevertheless, the patch location and shape models had very dissimilar capabilities in explaining the data variability, indicating that it was possible that some patch features were more greatly dependent on the disturbances considered than other features. The patch location model had a very good fit to the data and a high coefficient of determination when compared to the average explanatory power in ecological research (Low-Décarie et al., 2014). Moreover, the model showed that the patch location was almost exclusively driven by the groundwater hydrology. In fact, the removal of the morphodynamic disturbance factor from the model did not result in a significant difference between the models (H_0: model and sub-model are not different, p-value=0.5569). The primacy of this driver was found for the patch location of each succession phase as well. Only the ES phase was noticeably negatively affected by morphodynamic disturbances, which appeared to be a key element for preventing vegetation encroachment by this succession phase. We must admit that we were fairly surprised with the outcome regarding the effect of morphodynamics on the very early succession phases, such as IP and PP, which we expected to be higher. The abrasion effect of suspended load and bed load along with floating debris is undeniably an important morphodynamic process that was neglected in these analyses, which could

explain the low impact of morphodynamics in very young succession phases. Likewise, shear stress, the proxy for morphodynamic disturbance, does not accurately represent the side erosion process, which is also an important effect, at least for meandering rivers (Scott et al., 1996). Nevertheless, we were persuaded to attribute this circumstance to the impact of extreme summer droughts that prevented the development of seedlings and younger succession phases in the river channel, in addition to a winter flow that inhibited the establishment and overtake by xerophyte species. Furthermore, this outcome is in consensus with the current knowledge that the physical habitat shaping in the inner river zones occupied by younger succession phases is controlled by lower and more frequent floods while older succession phases are bridled by higher and less frequent floods (Wolman and Miller, 1960), such as the maximum annual discharges used to characterize MDi. Accordingly, it appears that in our typically Mediterranean study sites, the location of the succession phase patches are more a result of zonation driven by water scarcity than ecological succession driven by different fluvial disturbances. This may probably be a common characteristic of Mediterranean-climate regions, where seasonal water scarcity is common and riparian ecosystems have similar adaptations and strategies to cope with the particular stresses of these analogous med-rivers (Bonada and Resh, 2013). Given this generalization, this circumstance seems to be one more particularity of Mediterranean riparian woodlands, because in temperate and tropical river systems the landform processes are considered to be the main driving factor of riparian vegetation (see Stromberg et al., 1996; Scott et al., 1999; Shafroth et al., 2000; Steiger et al., 2005; Stromberg et al., 2007a). Although, other factors such as river geomorphology may prompt this divergence. Actually, all of the case studies considered were in a valley confined to some extent, with bed material ranging from bedrock to cobles, which may preclude channel movement and sediment transport, resulting in a much more stable river channel with low sinuosity in which broad processes of erosion/sedimentation occur with much more difficulty (Dingman, 2009). Soil composition has also been pointed as determinant in riparian landscape characterization (e.g., Molina et al., 2004) but in our case no linear relation was found between bed substrate and succession phases or their location. Furthermore, we considered substrate to be a result of fluvial disturbance rather than a component of it. For these reasons, we did not include bed substrate as a latent variable in our models. Nevertheless, the outcome of this analysis

may still serve conservation and management purposes. In addition to the creation of the necessary morphodynamic disturbance to prevent vegetation encroachment and to preserve the naturalness of the riparian landscape, particularly in the younger succession phases, such results point to the necessity of maintaining a minimum river discharge capable of sustaining water table levels. Such sustenance can in fact prevent the rearrangement of the riparian succession phases following the increased water stress imposed by river regulation. Furthermore, it may also help in preventing the invasion by exotic species, as those appear to be able to adjust faster to new disturbance regimes than native species (Planty-Tabacchi et al., 1996). But in the end, the findings of this work are consistent with observations in previous studies in Mediterranean climate rivers, where changes in water table levels determined the rearrangement of the riparian communities according to the hydrologic thresholds of the species (e.g., Stromberg et al., 1996; Scott et al., 1999; Shafroth et al., 2000; Stromberg et al., 2007a). Notwithstanding, the novelty of this study was the quantification of the importance of each fluvial disturbance in the landscape features of riparian vegetation.

As for the shape of the patches, groundwater hydrology is still its major driver, even though morphodynamic disturbances were shown to have a much more important role, where greater distances to the water table elevation account for larger and less complex patches. However, the disturbances reported cannot be considered to be substantial drivers of this feature, as indicated by the very low explanatory power of this model. Although common in ecological models (Møller and Jennions, 2002), this indicates that the majority of the patch shape variation comes from the residual variance of the model, suggesting that other factors are more influential in riparian patch shaping. Notwithstanding, there is still no consensus about the main drivers of this feature. Other factors such as geomorphology, human impact, valley width or stream sinuosity have been implicated in previous studies (e.g., Fernandes et al., 2011; Aguiar et al., 2016; Zhou et al., 2016), although not using a confirmatory factor analysis, but which can certainly be effective driving forces of patch complexity.

In conclusion, fluvial disturbances were demonstrated to have different effects on the location and shape of riparian vegetation patches. The main driver of riparian patch location was groundwater hydrology and indicated the predominant zonation of riparian succession phases over natural ecological succession. Nevertheless, morphodynamic

disturbances are still responsible for preventing vegetation encroachment. However, patch shape seemed not to be primarily driven by fluvial disturbances but, within the limited explained variability of the proposed model, both groundwater hydrology and morphodynamic disturbances had a substantial impact. These outcomes emphasize the likely necessity for specific procedures during flow regime management to account for the particularities of the drivers of fluvial disturbances of riparian vegetation in Mediterranean rivers.

SECTION IV

RIPARIAN VEGETATION MANAGEMENT

CHAPTER 5

Reducing river regulation effects on riparian vegetation using flushing flow regimes

Rivaes, R., Rodríguez-González, P. M., Albuquerque, A., Pinheiro, A. N., Egger, G., Ferreira, M. T. (2015). Reducing river regulation effects on riparian vegetation using flushing flow regimes. *Ecological Engineering* 81:428-438. DOI: 10.1016/j.ecoleng.2015.04.059

Resumo

Uma das causas mais relevantes da degradação dos sistemas dulçaquícolas é a alteração física do habitat provocada pelo represamento dos rios. Os caudais ambientais reduzem essa degradação mas continuam na generalidade fundamentados nos requisitos de espécies aquáticas, desatendendo outros componentes bióticos do ecossistema, tal como a vegetação ripária. Mesmo assim, quando os métodos de caudais ambientais alegam ter em consideração os habitats ripários e propõem caudais específicos para tal, os seus resultados raramente são prognosticados de forma quantitativa previamente à sua implementação. Utilizou-se um modelo dinâmico de vegetação ripária para analisar a dinâmica do mosaico ripário predito para diferentes regimes de caudal em dois troços de rio, bem como para avaliar os requisitos da vegetação de forma a assegurar a manutenção ecológica e vitalidade a longo prazo da estrutura ripária em rios com regimes de caudais modificados. Para além disso, analisou-se a capacidade dos caudais de limpeza em restaurar e gerir a vegetação ripária, assim como a eficiência dos caudais ambientais em satisfazer os requisitos de vegetação ripária. Mostrou-se que a invasão do canal por parte da vegetação ripária é evitada principalmente por cheias com período de retorno de pelo menos dois anos mas que o planeamento do regime de caudais ambientais focados em atender tais requisitos de vegetação ripária é específico da bacia hidrográfica. Para além disso, os caudais de barragem controlaram a invasão do canal pela vegetação sem causar impactos geomórficos severos no leito do rio a jusante e com pequenas perdas de água para os gestores das barragens.

Palavras-chave: caudais ambientais, vegetação ripária, regularização fluvial, restauro fluvial, caudais de limpeza, gestão de barragens

Abstract

One of the most salient causes of the degradation of freshwater systems is the physical habitat changes attributed to river damming. Environmental flows reduce such degradation but are still generally based on the requirements of aquatic species and disregard other biotic components of the ecosystem, such as riparian vegetation. Nevertheless, when environmental flow methods claim to consider riparian vegetation habitats and propose specific flows, their outcomes are rarely predicted quantitatively prior to their implementation. We used a dynamic floodplain vegetation model to analyze the riparian patch dynamics predicted for different flow regimes in two river stretches and to assess vegetation requirements to ensure long-term ecological maintenance and vitality of riparian structures in rivers with altered flow regimes. Furthermore, we assessed the capability of flushing flows to restore and manage riparian vegetation and the efficiency of environmental flows to satisfy riparian vegetation requirements. We found that vegetation encroachment is mainly prevented by floods with a recurrence interval of at least 2 years but that environmental flow regime planning aimed at complying with riparian vegetation requirements is watershed-specific. Additionally, reservoir flows controlled vegetation encroachment without causing severe geomorphic impacts on downstream river channels and with minor water losses to dam managers.

REDUCING RIVER REGULATION EFFECTS ON RIPARIAN VEGETATION USING FLUSHING FLOW REGIMES

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Author contributions

Conceived and designed the experiments: RR PRG AA MTF. Performed the experiments: RR PRG AA. Analyzed the data: RR PRG AA AP MTF. Contributed reagents/materials/analysis tools: RR AP. Wrote the paper: RR PR AP MTF.

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1. Introduction

Dams are built to retain water for human needs. However, dams also create environmental consequences, such as ecological responses to instream and riparian species proportional to the alteration of flow regime (Poff and Zimmerman, 2010). Additionally, modifications in river flow regimes are expected to increase due to increased water withdrawals to satisfy human need (Alcamo et al., 2007a; Vörösmarty et al., 2010; Murray et al., 2012), and freshwater systems will endure greater biodiversity losses than other ecosystems, particularly Mediterranean ecosystems (Sala et al., 2000). The contradictory urge to protect river environments while satisfying human water demand remains one of the most important challenges of our time (Nilsson and Berggren, 2000; Palmer, 2010). Environmental flow management has been an ongoing scientific issue for the last two decades (e.g., Arthington and Zalucki, 1998; Dyson et al., 2003; Hughes and Rood, 2003; King et al., 2003; Acreman and Dunbar, 2004; Rood et al., 2005; King and Brown, 2006; Poff et al., 2010), prompting the use of an extensive number of methods to determine environmental flow. The most advanced methods for determining environmental flow requirements maintain biological community functions and riverine processes while focusing on flow regime over space and temporal scales. Nevertheless, these methods are generally based on the requirements of aquatic species, mostly fish (Acreman et al., 2009), and usually disregard other biotic and abiotic components. Such approaches set aside the inter-annual flow variability that rules longer lifecycles, bypassing an important aspect in river management: the long-term perspective of the riverine ecosystem (Stromberg et al., 2010b).

Riparian species and communities are suitable environmental change indicators (Nilsson and Berggren, 2000; Rodríguez-González et al., 2014) and should be considered in the development of environmental flow regimes. The succession dynamics of riparian vegetation is a medium- to long-term active process that responds directly to flow regime and its disturbance (Junk et al., 1989; Poff et al., 1997; Richter et al., 1997; Toner and Keddy, 1997; Mallik and Richardson, 2009). Furthermore, riparian vegetation is important in aquatic habitat improvement (Naiman and Décamps, 1997; Naiman et al., 2005; Ghermandi et al., 2009) and biological conservation (Broadmeadow and Nisbet, 2004; Van Looy et al., 2013).

Several studies describe the response of riparian vegetation to flow regime changes (e.g., Greet et al., 2011a; Greet et al., 2011b; Johnson and Waller, 2012; Miller et al., 2013; Webb et al., 2013). However, few studies have used a spatially detailed approach to riparian patches to examine the responses of riparian vegetation (e.g., Benjankar et al., 2012; Egger et al., 2012; García-Arias et al., 2013; Rivaes et al., 2013). These cause-and-effect relationships between flow regime and riparian vegetation have seldom been used in regulated river management (e.g., to prevent vegetation encroachment and maintain sustainable riparian landscapes). These cause-and-effect relationships are important in Mediterranean climates where the downstream flow regulation of dams is acute and widespread and where hydrologic-driven changes on vegetation are most likely stronger because flow regulation persists for longer river stretches downstream of dams (Bejarano and Sordo-Ward, 2011). Additionally, as a result of regulation, base flows may increase and provide water throughout the growing season (i.e., reversing the natural flow regime), favoring the development of later successional vegetation stages and ultimately reducing biodiversity (Magdaleno and Fernández, 2010).

Reservoir outflows, in the form of flushing flows, are intended to mimic the effects of natural flows toward the removal of fine sediment and channel maintenance (Kondolf, 1998), encompassing the scour of undesired vegetation (Milhous, 2012), which is of great importance to channel flow conveyance. Methods with pre-defined flushing flows concerning this matter recommend discharges ranging from a certain percentage of the mean annual flow up to floods with a recurrence interval of 10 to 15 years. However, the majority of methods focus only on the sediment transport capacity of these flows, whereas rejuvenation of the riparian patch mosaic has been poorly analyzed.

This study simulated the riparian vegetation response to flow regime management to i) determine the riparian vegetation response to different flushing flows and determine which regime would be able to re-establish riparian patch dynamics, ii) evaluate the possibility of reducing the effects of flow regulation on riparian vegetation downstream of dams through reservoir management, iii) assess the potential damage of releasing sediment-deprived flushing flows on fluvial geomorphology, and iv) analyze the adequacy of existing general environmental flow guidelines for riparian vegetation.

2. Materials and methods

2.1. Study sites

Two study sites ("Monte da Rocha" and "Odelouca") were used to assess vegetation response to regulated flow regimes. The Monte da Rocha study site is located in southern Portugal (Lat. 37º 44' 11,75''N, Long. 8º 18' 04,23''W) near Panóias village, on the headwaters of the Sado River. Its flow is regulated by the Monte da Rocha dam, located approximately 1 km upstream. The Monte da Rocha dam is a 40-year-old irrigation infrastructure without any intentional flow release. Nonetheless, there is permanent low flow at the study site derived from the percolation through the dam body and foundation, which slightly increases during the crop irrigation period due to the operation of the pumping station. The riparian woody vegetation in the area is largely composed of willows (*Salix atrocinerea* Brot.), ashes (*Fraxinus angustifolia* Vahl) and tamarisks (*Tamarix africana* Poir). The characteristic terrestrial species in the uplands are cork oaks (*Quercus suber* L.) and holm oaks (*Quercus ilex* L. subsp. *ballota*).

The Odelouca study site is located a little further south (Lat: 37º 23' 05,00''N, Long: 8º 18' 39,46''W) in the upper course of the Odelouca River, upstream of the Odelouca Reservoir. The river here is free-flowing, and its riparian vegetation is close to natural. The flow regime is typically Mediterranean, having two distinct periods: the winter period, with low flows that are sporadically interrupted by flash floods, and the summer period, with very low or even null flows. This river stretch was selected as the best available site with near natural conditions in terms of riparian vegetation and is representative of the downstream Odelouca River. The composition of riparian vegetation here is similar to the first study site, with willows, tamarisks and ashes being the most common woody species (Figure 29).
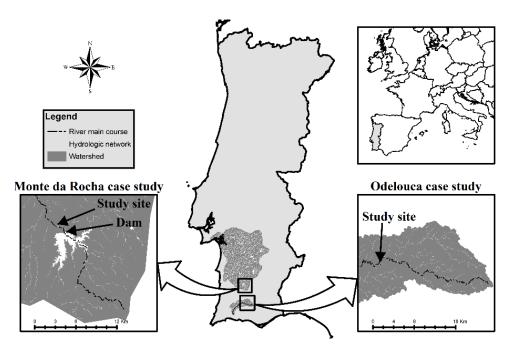
Figure 29. Study site location (Portugal in light grey and river watersheds in dark grey).

2.2. Vegetation model

We used the dynamic floodplain vegetation model *CASiMiR-vegetation* (Benjankar et al., 2011) to determine the expected response of riparian vegetation communities to flow regime changes. This is a physically based numerical model that includes empirical relationships between relevant hydrological characteristics (Poff et al., 1997) and the riparian guild level responses to permanent hydrologic regime changes (Merritt et al., 2010). Its outputs are spatially explicit vegetation maps of the riparian vegetation patches by succession phase. The *CASiMiR-vegetation* model has been used to examine Mediterranean climate conditions with good results, including one of the study sites presented in this study (see García-Arias et al., 2013, and Rivaes et al., 2013, for model structure, calibration and performance).

2.3. Input data

2.3.1. Hydrological and meteorological data

Hydrological and meteorological data were obtained from the National Water Resources Information System (SNIRH, 2010) and were used to create different input flow regimes for the riparian vegetation modeling. The natural flow regime at Monte da

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Rocha was estimated from hourly precipitation records obtained on several meteorological stations located in the drainage basin and nearby. Based on these data, natural maximum discharges at the study site were obtained for each year from the recorded precipitation using the USA Soil Conservation Service (SCS) unit hydrograph method (SCS, 1972). Peak discharges corresponding to specific recurrence periods were estimated according to the local corresponding IDF (Intensity-Duration-Frequency) curves (Brandão et al., 2001).

We used data for the Odelouca case study from a nearby gauging station (Monte dos Pachecos) to estimate site-specific maximum instantaneous discharges for different probabilities of exceedance. The annual maximum discharges at the study site were calculated considering the ratio between "Monte dos Pachecos" and the study site drainage basin areas and their mean annual precipitation.

2.3.2. Flow regime definition

Initially, three different flow regimes were considered for vegetation modeling: dam-operated, natural and pre-defined environmental flow (hereafter called Eflow) regimes (Table 8). The dam-operated flow regime is applicable to the Monte da Rocha study site and considers the regulated flow regime below the dam over the past decade. The dam-operated flow regime was used to produce a riparian vegetation map for the actual regulated river, thus enabling model calibration by comparing with the present vegetation patches. The calibrated model was used to produce vegetation maps of the expected patch disposal under a natural flow regime and to provide a benchmark for the evaluation of different flow regimes, considering riparian vegetation as the biotic indicator element. Pre-defined Eflow regimes were set according to the hydrologic method developed by Alves et al. (2003) because it is one of the few methods that includes pre-defined specific guidelines for riparian vegetation requirements in Mediterranean streams commonly used in Portugal. Furthermore, this method recommendation stands on the average flow values of the existing literature recommendations. The method is based on hydrological records and recommends a biannual release of a 2-year flood event intended for riparian vegetation and channel maintenance. However, the ecological basis of this method was never demonstrated and its effectiveness therefore remains doubtful.

Table 8. Considered flow regimes during vegetation modeling. Values stand for annual maximum discharges (m³/s).

Monte da Rocha case study				Odelouca case study		
Year	Dam-operated	Natural	Eflow	Year	Natural	Eflow
1999/2000	0.03	80	0.4	**1984/1985**	283	0.7
2000/2001	0.03	153	313	**1985/1996**	146	122
2001/2002	0.03	147	0.4	**1986/1987**	455	0.7
2002/2003	0.03	143	313	**1987/1988**	292	122
2003/2004	0.03	341	0.4	**1988/1989**	284	0.7
2004/2005	0.03	181	313	**1989/1990**	35	122
2005/2006	0.03	194	0.4	**1990/1991**	1	0.7
2006/2007	0.03	339	313	**1991/1992**	27	122
2007/2008	0.03	219	0.4	**1992/1993**	22	0.7
2008/2009	0.03	204	313	**1993/1994**	1	122
2009/2010	5	166	0.4	**1994/1995**	170	0.7

2.3.3. Hydraulic data

The hydraulic data required by the *CASiMiR-vegetation* model is the maximum annual discharge shear stress and the minimum annual water surface elevation. *HEC-RAS 4.1.0* (Brunner, 2008) was used to determine the flow rating curve at the study sites outflow sections, and the *River2D v0.93* (Steffler et al., 2002) was used to predict shear stress and water surface elevation for each of the considered discharges. For both models, hydrodynamic calibration was achieved by changing vegetation roughness to attain the best possible agreement between the modeled and the measured flow depths for different discharges (Appendix B). Data were processed with *ArcGis™ 9.2* (Environmental Systems Research Institute, 2010) to comply with the vegetation model input data file requirements.

2.3.4. Vegetation data

Vegetation surveys were performed at each study site to support model calibration and provide vegetation maps for the assessment of model accuracy. For the different vegetation patches, species and habitat characteristics were recorded and used to

define ecological succession phases to enable the calibration of the model vegetation-related parameters. Additional information on this issue is provided by (Egger et al., 2013). Five similar succession phases were found in both case studies: the Initial Phase (IP), Pioneer Phase (PP), Early Successional Woodland Phase (ES), Established Forest Phase (EF) and Mature Forest Phase (MF). IP is characterized by sand or gravel open bars with less than 50% vegetation cover and the absence of woody arboreal species. PP is typified by the dominance of woody arboreal species recruitment and ES is characterized by the presence of well-established pioneer individuals, mainly microphanerophyte (Raunkiaer, 1934) pioneer species, such as willows and tamarisks. EF is found in patches presenting high canopy cover of mesophanerophyte (Raunkiaer, 1934) species, such as ashes, along with a well-defined understory stratum. The Mature Forest is characterized by a vegetation cover similar to EF but also contains terrestrial arboreal cork oak or holm oak species, which are the typical upland vegetation of these Mediterranean landscapes.

2.4. *Vegetation modeling*

Vegetation modeling was performed with site-specific calibrated versions of the *CASiMiR-vegetation* model based on the vegetation surveys used to sustain the parameterization of the age and height of the succession phase above the water table. García-Arias et al. (2013) and Rivaes et al. (2013) presented detailed descriptions of the model rationale and parameter calibration. The initial vegetation map was provided by the model's Initial Condition module. This module provides a riparian landscape that represents a hypothetical arrangement of riparian succession phases according to height above water table ranges and therefore indirectly incorporates the historical fluvial disturbance that characterizes the mid- and long-term habitat conditions at this river reach. Vegetation evolution was modeled along 10-year periods ending in the years when the vegetation was surveyed, considering the corresponding series of maximum annual discharges, to compare obtained and observed vegetation maps. The 10-year periods were used to avoid the effect of the initial vegetation conditions on the results, which influence only the first five years of simulation. Longer simulation periods were not considered due to the relevance that river morphological changes may assume in vegetation development (Politti et al., 2014). Nonetheless, a sediment

transport analysis was made for the 10-year period to check if sediment transport could play a significant role in the study sites (subchapter 2.6).

At the Monte da Rocha site, model accuracy was assessed by comparing the observed and expected vegetation maps, considering the present flow regime and using different statistics. In this paper, model accuracy is only assessed for the Monte da Rocha study site because the results from the model calibration and validation for Odelouca had already been published (García-Arias et al., 2013; Rivaes et al., 2013).

For both case studies, expected riparian vegetation maps resulting from flow regime scenarios were compared with the respective expected natural vegetation. Furthermore, we tested the existence of significant differences on the mean area balance of each scenario using a Wilcoxon-Mann-Whitney test following testing for normal distribution using a Shapiro-Wilk normality test and the R stats package in R language (R Development Core Team, 2011).

2.5. Flushing flow regime analysis

Wherever the considered Eflow regime was unable to re-establish naturalized riparian vegetation, diverse regimes of flushing flows with different recurrence intervals (recreating large and intermediate floods) were tested (Table 9). This procedure allows us to determine the most efficient flooding regime concerning riparian vegetation to preserve the river morphological structure and maintain a riparian patch mosaic as close as possible to the reference condition.

Table 9. Flushing flow regimes considered in the Odelouca flushing flow regime analysis.

	Recurrence interval													
Main flood	Regular flood	3 year return	5 year return	10 year return	2 year return	3 year return	3 year return	5 year return	5 year return	5 year return	10 year return	10 year return	10 year return	10 year return
Intermediate flood	None	None	None	None	Regular flood	Regular flood	2 year return	Regular flood	2 year return	3 year return	Regular flood	2 year return	3 year return	5 year return
Year	**Maximum annual discharge (m³/s)**													
1	80	0.7	0.7	0.7	80	80	0.7	80	0.7	0.7	80	0.7	0.7	0.7
2	80	0.7	0.7	0.7	122	80	122	80	122	0.7	80	122	0.7	0.7
3	80	171	0.7	0.7	80	171	171	80	0.7	171	80	0.7	171	0.7
4	80	0.7	0.7	0.7	122	80	0.7	80	122	0.7	80	122	0.7	0.7
5	80	0.7	225	0.7	80	80	122	225	225	225	80	0.7	0.7	225
6	80	171	0.7	0.7	122	171	171	80	0.7	0.7	80	122	171	0.7
7	80	0.7	0.7	0.7	80	80	0.7	80	122	0.7	80	0.7	0.7	0.7
8	80	0.7	0.7	0.7	122	80	122	80	0.7	171	80	122	0.7	0.7
9	80	171	0.7	0.7	80	171	171	80	122	0.7	80	0.7	171	0.7
10	80	0.7	225	290	122	80	0.7	225	225	225	290	290	290	290
11	80	0.7	0.7	0.7	80	80	122	80	0.7	0.7	80	0.7	0.7	0.7

Expected vegetation maps resulting from the simulated flow regimes were compared with the expected natural vegetation map. Changes were assessed with Cohen's Kappa (Cohen, 1960), considered a valuable tool for assessing the accuracy of the model (Benjankar et al., 2010), fuzzy kappa (Visser and de Nijs, 2006) and the Root Mean Square Error (RMSE) of the covered area of succession phases. The first two statistics are measures of inter-rater agreement for categorical classifications and are most appropriate for directly accounting for a pixel-by-pixel comparison, attaining higher scores for more overlapping predictions and observations. RMSE measures the existing error between predicted and observed succession phase areas during the entire modeling period, presenting lower values for more accurate classifications.

2.6. Sediment transport analysis

In both case studies, the sediment transport capacity of the most appropriate flushing flows for riparian vegetation maintenance was analyzed to search for possible issues of excessive erosion or sedimentation at the study sites caused by the proposed flushing flows. The sediment analysis module of the *HEC-RAS 4.1.0* model (Brunner, 2008) was used for this purpose. *Quasi-unsteady flow* data considered site-specific flooding periods linked to the watershed time of concentration following a flood wave computed by the SCS unit hydrograph method (SCS, 1972). Initial conditions and sediment transport parameters were determined according to the existing bed gradation based on soil texture analysis. The "hungry water" phenomenon occurring downstream of the dam (Kondolf, 1997) was simulated using a boundary condition with a null sediment load series. The boundary condition details are presented in Appendix B.

3. Results

3.1. Model calibration and vegetation modeling

Model calibration for the Monte da Rocha study site achieved a quadratic weighted kappa of 0.69 when observed as a proportion of the maximum possible given the observed marginal frequencies, and fuzzy kappa reached a value of 0.49. Detailed

information about parameter calibration and the validation confusion matrix is presented in Appendix C. These results are considered to be of a moderate to good strength of agreement between observed and expected vegetation maps (Landis and Koch, 1977; Altman, 1991; Viera and Garrett, 2005) and are found at the same rank of strength agreement as the calibration and validation results attained for Odelouca by Rivaes et al. (2013).

At Monte da Rocha, the three tested flow regimes resulted in diverse patch area combinations, largely in the bank zone. The floodplain succession vegetation (EF and MF) was not affected by the imposed discharges. In the natural flow regime scenario, IP occupies more than a quarter of the study area (near 29%), reaching nearly 34% of the surface in some years and never less than 12% of the surface. Furthermore, this succession phase is highly responsive to discharge magnitude, and its area variation reflects the inter-annual discharge balance. Pioneer vegetation (PP) is capable of settling on the channel surface on a regular basis, although with much less cover (never occupying more than 7% of total area). This succession phase is highly related to flood events and colonizes areas once occupied by IP during floods. In the natural flow regime scenario, ES is retrogressed to a stable condition, where it occupies nearly 8% of the total area and appears to be determined only by major flood events that prevent the invasion of the active river channel by this phase. EF and MF present very stable areas that are undisturbed by the studied flood events and account for approximately 38% and 20% of the total area, respectively (Figure 30– Natural).

In the present dam-operated flow regime, vegetation encroachment in the river channel is present, and the last flood event corresponding to a regular flood was unable to retrogress it. IP never achieves more than 11% of total area, and younger succession phases are able to succeed toward ES, which occupies the entire active river channel and represents up to 30% of the study area. Such an evolution represents an approximately 300% area increase of the ES when compared to the natural flow regime scenario. PP is also slightly increased, whereas EF and MF remain untouched in the floodplain zone to the same extent as the natural flow regime (Figure 30 – Dam-operated).

Considering the pre-defined Eflow regime, the succession phases reach similar areas to those of the natural flow regime. In such cases, IP occupies nearly 30% of the total

area, whereas PP and ES occupy approximately 3% and 9%, respectively. Once again, EF and MF are unaffected by the imposed floods and maintain their areas. The results of comparing EF and MF with the natural flow regime vegetation map show only a slight increase of ES of approximately 1%, whereas IP and PP increase approximately 1% and 2% (Figure 30 – Eflow).
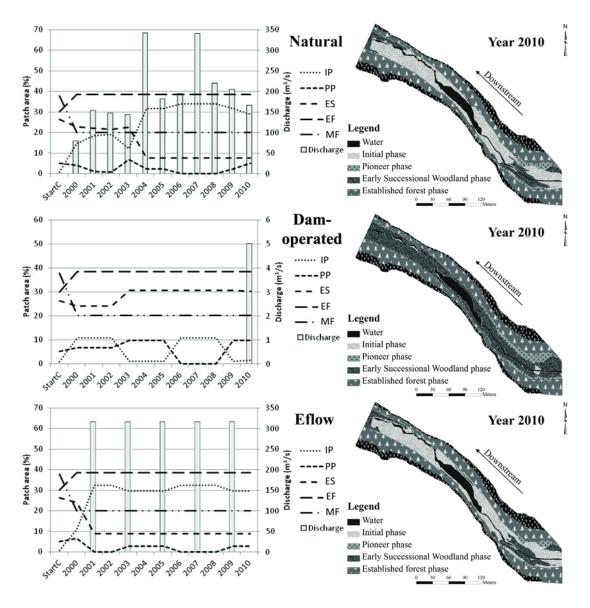
Figure 30. Riparian patch dynamics in Monte da Rocha study site according to three different flow regimes. From top to bottom: natural, dam-operated and environmental flow regimes. Line plots represent riparian succession phases evolution consistent with the annual maximum discharges presented in the bar plots. Study site maps stand for the 2010 expected vegetation maps according to each flow regime.

As for the Odelouca case study, the expected vegetation map for the natural flow regime shows a riparian mosaic where IP covers approximately 35% of the total area, matching the mean area value of this phase for the entire modeling period. PP is also very variable between years, ranging from 2% to 31% along the modeling period. Similar to the previous case study, these phases are highly responsive to flood events, ceding ground to one another according to the existing flood disturbance. ES and EF are not frequently disturbed and cover approximately 21% and 38% of the total area, respectively. Nevertheless, similar in the previous case study, it is clear that major flood events restrict channel invasion by ES (Figure 31 – Natural).
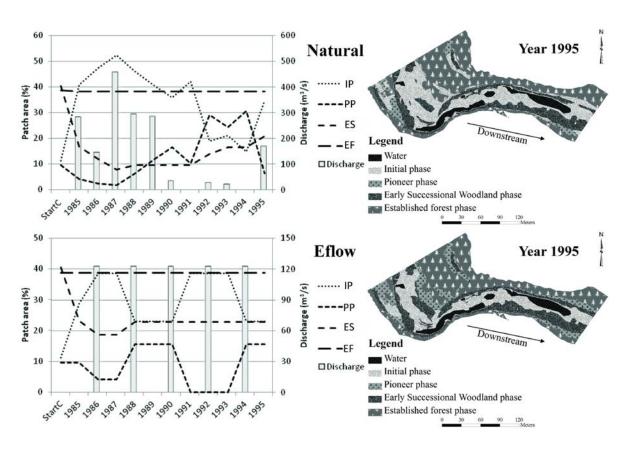
Figure 31. Riparian patch dynamics in Odelouca study site according to natural (top) and environmental (botton) flow regimes. Line plots represent riparian succession phases evolution consistent with the annual maximum discharges presented in the bar plots. Maps stand for the 1995 expected vegetation map for the correspondent flow regimes.

Compared to the expected natural vegetation map, a regulation scenario with the considered pre-defined Eflow regime causes a different expected patch mosaic. In this case, IP remains at nearly 23%, and PP reaches 16% due to decreased disturbance

and consequent settlement of pioneer vegetation on the river channel. This outcome shows a swop of approximately 10% of total area in between those phases when compared to the former scenario. In turn, ES stabilizes at approximately 22% of the total area, and EF maintains the same 39% end-to-end with negligible differences compared to the former scenario (Figure 31 – Eflow).

The comparison analysis of the succession phase's area development throughout the entire modeling period revealed that the covered area of succession phases driven by the natural and Eflow regimes at Monte da Rocha are not significantly different. However, for the IP, PP and ES, both regimes differ significantly from the dam-operated regimes. At the Odelouca study site, the comparison between natural and Eflow regimes exposes significantly different areas for the two older succession phases (ES and EF) but not for the younger succession phases, namely, IP and PP (Figure 32).
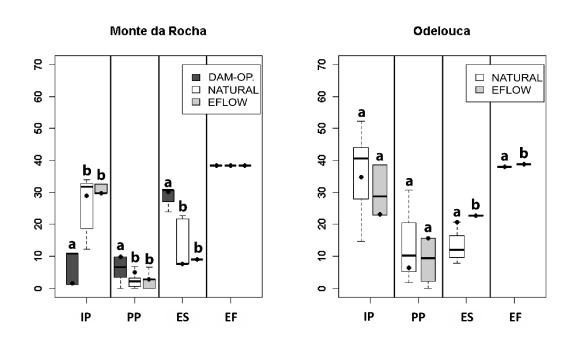
Figure 32. Area cover variation of succession phases across the 10-year modeling period consistent with the considered flow regimes in both case studies. Wiskers, boxes and thick lines stand for non outlier maximums, first and third quartiles, and average, respectively. Black dots represent cover areas for the considered expected vegetation maps in the last modeling year. Letters stand for significant groups of flow regimes in each succession phase.

3.2. Flushing flows

According to the kappa statistic, the best flow regime incorporating flushing events for Odelouca is composed of 10-year floods interspersed with 3-year floods. Such flow regime achieved fourth place according to fuzzy kappa. Detailing such a statistic by succession phase shows that this flow regime attains the best fuzzy kappa results of all of the considered flow regimes in three out of four succession phases. RMSE parsing also sustains this outcome because this flow regime is among the top five flow regimes showing the least area errors in succession phases throughout the entire modeling period (Figure 33).

Figure 33. Agreement evaluation between expected vegetation maps resulting from different flush flow regimes and the natural expected vegetation map of Odelouca study site. Agreement was appraised with Cohen's kappa (left), fuzzy kappa (center) and succession phase area's Root Mean Square Error for the 10-year modeling period (right).

3.3.   Sediment transport analysis

The proposed flushing flow regimes exerted similar geomorphologic effects on both case studies. The 2-year flood event at Monte da Rocha induced a mean net topographic change of minus 4 mm. The river channel is dominated by erosion, which overcomes sedimentation by 20%, and 46% of the river channel remained unchanged. Nevertheless, topography changes never exceed 29 mm of erosion (more than half of the erosion is lower than 5mm) or 3 mm of sedimentation (where 46% of the sedimentation is less than 2mm).

In the Odelouca case study, erosion is the predominant phenomenon occurring in both floods. The mean net topographic change caused by the intermediate 3-year flood events is a 22-mm decrease. Herein, erosion is expected in 57% of the river channel, and only 32% is predicted to aggrade. Erosion never goes beyond 193 mm deep (nearly 80% of the erosion is less than 16 mm) and sediment deposition is always lower than 19 mm (70% of the deposition is less than 6 mm). Similarly, predictions for the main 10-year flood event show that 68% of the riverbed will erode and only 25% is expected to suffer sediment aggradation. Mean net topographic change is minus 32 mm, ranging from 221 mm of erosion to 11 mm of aggradation. Nevertheless, in 61% of the eroded area, erosion is less than 21 mm and aggradation is less than 3 mm in 69% of the sedimentation area (Figure 34).
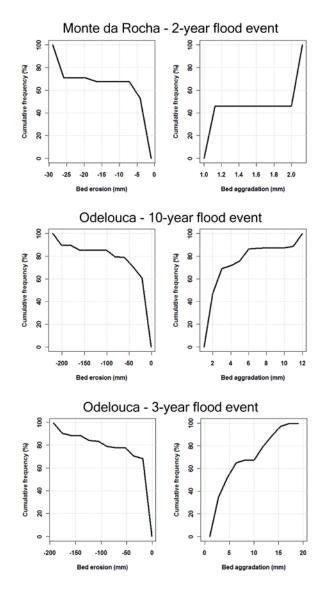
Figure 34. Cumulative frequencies of expected erosion and aggradation phenomena originated by the proposed flush flows on the study sites.

4. Discussion and conclusions

The evolution of vegetation patches was simulated using *CASiMiR-vegetation* model, which was successfully implemented in both regulated and unregulated study sites with considerable good calibration and validation results. The numerical model results showed that the natural flow regime brings much more variability to the fluvial patch mosaic than any of the other regulated regimes. This circumstance can be attributed to the great inter-annual variability of natural Mediterranean rivers, which is of extreme importance to the organisms and population dynamics within riparian vegetation (Pettit

et al., 2001; Stella et al., 2013). Flood events with recurrence intervals of at least two years are primarily responsible for retrogressing vegetated patches and for preventing vegetation encroachment inside the channel. This condition is evident not only by the response of succession phases to different flood events (as in natural and environmental flow regimes) but also in flow regimes with low variability (dam-operated flow regime at Monte da Rocha), where vegetation encroachment was able to settle in less than a decade. This prediction is coherent with studies where riparian forests were monitored to evaluate their response to flow regulation downstream of a dam (e.g., Shafroth et al., 2002; Braatne et al., 2007; Bejarano et al., 2011; Bejarano and Sordo-Ward, 2011; Benjankar et al., 2012; Egger et al., 2012) and demonstrates the capacity of *CASiMiR-vegetation* to predict riparian responses to flow regime management, as well as the benefit of including riparian communities in environmental flow approaches.

The dam-operated flow regime at Monte da Rocha causes the most important divergence from the natural state condition and allows vegetation succession to mature phases inside the channel. This tendency is supported by a consistent significant difference in the area cover of succession phases between this flow regime and the other flow regimes.

In both case studies, the area variation of succession phases caused by the Eflow regime overlaps substantially with the succession phases originated by the natural regime. However, the smaller flow variation always creates a smaller inter-annual variation in the riparian patch mosaic. This Eflow regime at Monte da Rocha seems to meet two conditions: it is enough to re-establish naturalized riparian vegetation and is the minimum acceptable for preventing vegetation encroachment. However, precipitation intensities in this region are close to Portuguese records (Brandão et al., 2001), which means that we have relatively significant discharges for low-return periods. In areas where precipitation intensities are comparatively lower, it may be necessary to consider discharges with higher return periods to guarantee the reinstatement of naturalized succession phases.

In contrast, the same method applied to the Odelouca site (a biannual release of a 2-year discharge) shows a noticeable reduction of patch retrogression and a stable succession of vegetation to mature stages in response to the considered Eflow regime. The mature vegetation (ES and EF) encroachment is evidenced by the significantly

different succession area ranges (Appendix D includes additional information on statistical analysis). Younger succession phase areas, namely IP and PP, are not significantly different between scenarios, but the expected vegetation maps resulting from a decade of regulation reveal succession phase areas that are approximately 12% and 9% farther apart from the correspondent succession phases in the natural flow regime. According to these results, the designed Eflow regime applied to Odelouca would be inadequate to maintain the natural patches of riparian vegetation and, therefore, instream ecosystem integrity.

The flushing flow regime analysis in the Odelouca case study points to a specific flow regime composed of a main 10-year discharge interspersed with 3-year discharges. Accordingly, discharges with lower recurrence intervals can control the initial succession phases, and longer recurrence intervals are required to maintain the natural patch mosaic of older phases. Such flood frequency corresponds to the artificial creation of the different metastable, oscillation and acyclic process types resulting from the impact of disturbances on vegetation succession and retrogression in natural rivers (Formann et al., 2013). Additionally, this flow regime includes small floods that recharge the water table and control the plant distribution and abundance, as well as large floods that shape the floodplain physical habitat and remove exotic species (Richter and Thomas, 2007). Such outcomes are plausible given that environmental flows can play an effective role in retracting encroachment (Miller et al., 2013) and that flood discharges have also been acknowledged to influence vegetation, as predicted by the *CASiMiR-vegetation* model (Scott et al., 1997; Friedman and Lee, 2002; Braatne et al., 2007; Richter and Thomas, 2007; Peake et al., 2011; Ibáñez et al., 2012). For these reasons, such an approach appears to be a good solution for linking riparian vegetation and river restoration (Gumiero et al., 2013).

Despite the sediment deficit created downstream of the dams, which may lead to riverbed incision (Rollet et al., 2013), the sediment transport analysis showed that the considered flushing discharges are not expected to cause severe geomorphic impacts. Although erosion is the predominant process occurring in both case studies, over a decade, only a few centimeters of mean net topographic change are expected to occur due to the proposed floods. Actually, some studies on river sediment transport suggest that major changes in sediment removal are due to lower magnitude but higher frequency floods occurring once or twice a year (Wolman and Miller, 1960; Nolan et

al., 1987). Nonetheless, the proposed floods entrain accumulated fine sediment, thus contributing to channel maintenance and to the physical habitat of aquatic communities (e.g., Sullivan and Watzin, 2010).

Additionally, the implementation of such flushing flows has implications for dam design and reservoir management. The Monte da Rocha reservoir has a gross capacity of 104.5 hm^3 and an effective storage of 99.5 hm^3. The bottom outlet maximum capacity is 80 m^3/s and the uncontrolled shaft spillway can release a maximum flow of 260 m^3/s. The 2-year event flushing flow proposed in the Eflow regime stands for a reservoir outflow of approximately 9 hm^3 with a maximum discharge of 313 m^3/s. The implementation of such an environmental flow regime would correspond to an annual flow release of 4.5 hm^3, meaning approximately 4.5% of the reservoir's effective storage.

In the Odelouca case, the reservoir has an average annual inflow of 122 hm^3 and a gross capacity of 157 hm^3, from which 134 hm^3 are effective storage. The gated chute type spillway has a maximum capacity of 1400 m^3/s, whereas the bottom outlet has a discharge capacity of 53 m^3/s. In this case, the considered flushing flows (10-year recurrence interval discharges interspersed with 3-year recurrence interval discharges) would represent releases of 9.9 hm^3 and 4.2 hm^3, respectively. However, the mean annual release corresponding to the proposed environmental regime will be 2.3 hm^3, which corresponds to only 1.7% of the reservoir's useful capacity, which, in turn, is similar to the mean annual inflow.

Thus, while the recommended flow regimes do not represent an important water loss, its applicability is highly hindered by the characteristics of the existing outlet structures of each dam. For Monte da Rocha, the lack of capacity of the bottom outlet to release the proposed flows, together with the absence of a gated spillway, renders it impossible to deliberately produce the specified flood discharges. The implementation of a modified Eflow, restricted to the current maximum capacity of the dam's bottom outlet, would be inadequate. The simulation of this scenario shows that such discharge is unable to prevent vegetation encroachment, resulting in a patch mosaic very similar to the existing patch mosaic.

Despite the bottom outlet lack of capacity for such discharges at the Odelouca dam, the gated spillway can release the water if the reservoir level is above the crest of the

spillway. These two cases demonstrate that the design of the dam outlet structures should consider the Eflows necessary for riparian vegetation maintenance, which are significantly larger than the most common Eflows for fish passage operation and fish habitat maintenance.

In sum, this study shows that adequate flushing flows have the ability to restore riparian patch dynamics and can reduce regulation effects on riparian communities downstream of dams. The proposed flushing flows did not cause effective damage on fluvial geomorphology. However, existing general environmental flow guidelines involve active water management and cannot be based merely on rules of thumb (Acreman and Dunbar, 2004).

The various effects of similar flow regimes on both case studies indicate that the establishment of guidelines for environmental flow regimes considering riparian requirements should only be applicable at the watershed scale (McCluney et al., 2014). Careful planning of environmental flows using a holistic perspective encompassing the ecological quality of regulated rivers, which should also include riparian requirements, must be based on a numerical modeling approach, preferably preceding dam design, so that the dam outlet structures can meet those requirements.

CHAPTER 6

Flow management to control excessive growth of macrophytes – an assessment based on habitat suitability modeling

Paper published in *Frontiers in Plant Science*

Resumo

Os rios mediterrânicos localizados em bacias hidrográficas intensamente agricultadas apresentam frequentemente crescimentos excessivos de plantas macrófitas aquáticas – particularmente espécies exóticas – devido à combinação entre altas concentrações de nutrientes na água e baixos caudais resultantes da subtração de água para irrigação. É habitual recorrer-se ao controlo mecânico e/ou químico para mitigar os problemas associados com o crescimento excessivo da biomassa vegetal, particularmente a diminuição da capacidade de vazão e o aumento do risco de cheias. Todavia, tais medidas de controlo são dispendiosas e laboriosas, não apresentando uma eficiência de longo prazo. Apesar de ser bem conhecida a grande sensibilidade da vegetação aquática às condições hidráulicas do escoamento, as abordagens de governação baseadas na gestão de caudais continuam relativamente inexploradas. O objetivo deste estudo foi, portanto, a de aplicar técnicas de simulação de habitat físico promovidas pelo método dos caudais incrementais (IFIM) a macrófitas aquáticas – a primeira vez a ser implementada neste contexto – de forma a modelar alterações na adequação do habitat sob diferentes cenários de caudal no rio Sorraia, centro de Portugal. Empregou-se esta abordagem com o intuito de testar se o risco de invasão e atafulhamento no canal por espécies inconvenientes poderia ser controlado pelo estabelecimento de caudais mínimos anuais. Foram usados 960 pontos de amostragem distribuídos aleatoriamente para examinar a adequação de habitat referente aos parâmetros hidráulicos de velocidade de escoamento, profundidade da água e dimensão do substrato, para as espécies locais mais relevantes, nomeadamente a espadana-de-água *Sparganium erectum*, a carvalha *Potamogeton crispus* e a invasora pinheira-de-água *Myriophyllum aquaticum*. Escolheu-se o período de menor caudal do ano para amostrar, de forma a analisar as condições hidráulicas no estado de perturbação mínima que possa permitir o estabelecimento e manutenção da vegetação. Posteriormente, recorreu-se ao modelo hidráulico bidimensional *River2D* para modelar a disponibilidade de habitat potencial em diferentes condições de escoamento, com base no índice de adequabilidade de habitat local específico para cada parâmetro hidráulico e espécie. Os resultados mostram que o crescimento e distribuição de plantas macrófitas aquáticas no período de perturbação mínima é resultado primeiramente das condições físicas do escoamento local. Usando curvas

de preferência específicas do local e um modelo bidimensional, foi possível determinar caudais anuais mínimos que poderão prevenir o crescimento excessivo e atafulhamento do canal originado pela espécie *Myriophyllum aquaticum*.

Abstract

Mediterranean rivers in intensive agricultural watersheds usually display outgrowths of macrophytes – notably alien species – due to a combination of high concentrations of nutrients in the water runoff and low flows resulting from water abstraction for irrigation. Standard mechanical and chemical control is used to mitigate the problems associated with excessive growth of plant biomass: mainly less drainage capacity and higher flood risk. However, such control measures are cost and labor-intensive and do not present long-term efficiency. Although the high sensitivity of aquatic vegetation to instream hydraulic conditions is well known, management approaches based on flow management remain relatively unexplored. The aim of our study was therefore to apply physical habitat simulation techniques promoted by the Instream Flow Incremental Method (IFIM) to aquatic macrophytes – the first time it has been applied in this context – in order to model shifts in habitat suitability under different flow scenarios in the Sorraia river in central Portugal. We used this approach to test whether the risk of invasion and channel encroachment by nuisance species can be controlled by setting minimum annual flows. We used 960 randomly distributed survey points to analyze the habitat suitability of the physical parameters 'flow velocity', 'water depth' and 'substrate size' for the most important aquatic species, including the invasive Brazilian milfoil *Myriophyllum aquaticum, Sparganium erectum* and *Potamogeton crispus.* We chose the lowest discharge period of the year in order to assess the hydraulic conditions while disturbances were at a low-point, thus allowing aquatic vegetation establishment and subsistence. We then used the two-dimensional hydraulic River2D software to model the potential habitat availability for different flow conditions based on the site-specific habitat suitability index for each physical parameter and species. Our results show that the growth and distribution of macrophytes in the hydrologically stable vegetation period is primarily a function of the local physical instream condition. Using site-specific preference curves and a two-dimensional hydraulic model, it was possible to determine minimum annual flows that might prevent the excessive growth and channel encroachment caused by *Myriophyllum aquaticum.*

Keywords: aquatic macrophytes, habitat suitability modelling, flow regulation, invasive species, *Myriophyllum aquaticum*, IFIM

FLOW MANAGEMENT TO CONTROL EXCESSIVE GROWTH OF MACROPHYTES – AN ASSESSMENT BASED ON HABITAT SUITABILITY MODELLING

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Authors contribution

KO, RR, TF, GE made substantial contributions to conception and design, and acquisition of data and analysis and interpretation of data. KO, RR, TF, GE participated in drafting the article and revising it critically for important intellectual content. KO, RR, TF, GE gave final approval of the version to be submitted. KO, RR, TF, GE take public responsibility for the content.

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1. Introduction

Aquatic macrophytes play an important role in riverine ecosystems, providing habitats for many organisms and affecting the hydraulic and chemical instream condition (Carpenter and Lodge, 1986). Their distribution and abundance are primarily determined by the hydrologic regime (frequency, duration and intensity of flood events) (Franklin et al., 2008; Riis and Biggs, 2003), which controls biomass loss and gain processes. Whereas loss processes are caused by increased drag forces during high flood events that cause stem breakage and uprooting of plants, biomass gain processes happen while disturbances are absent during medium to low flow conditions (Riis et al., 2008). In these stable interflood periods, macrophyte growth is controlled by several physical and chemical factors, including flow velocity and depth (Chambers et al., 1991; Riis and Biggs, 2003), light availability (Carr et al., 1997; Köhler et al., 2010), water temperature (Barko et al., 1986; Carr et al., 1997), and riverbed grain size (Baattrup-Pedersen and Riis, 1999), as well as the nutrient content of the riverbed and water (Barko et al., 1986; Demars and Edwards, 2009). Anthropogenic disturbances, such as high nutrient concentrations from water runoff (Jones et al., 2002; Mainstone and Parr, 2002), low suspended sediment concentrations and the resulting increase in light availability from river damming (Köhler et al., 2010; Madsen et al., 2001) and stabilization of the flow regime (less floods) (Franklin et al., 2008; Riis and Biggs, 2003) can alter the ecological equilibrium of the system and have been shown to stimulate excessive growth of aquatic vegetation, notably invasive alien species (Bunn and Arthington, 2002). This is known to cause various forms of ecological and economic damage (Brundu, 2014), including changes in species composition and richness (Bunn and Arthington, 2002; O'Hare et al., 2006), increased flood risk through higher flow resistance (Nikora et al., 2008; Vereecken et al., 2006), and interferences with human water uses such as water abstraction, hydropower, recreation and river navigation (Gómez et al., 2013; Halstead et al., 2003). Management of aquatic macrophytes by mechanical (cutting) or chemical (herbicides) means is therefore common practice in many rivers worldwide (Hussner et al., 2017; Madsen, 2000).

Especially in regulated Mediterranean rivers flowing through intensive agricultural watersheds and presenting prolonged spells of low flows the outgrowth of aquatic vegetation, and notably alien species, is a common phenomenon (Aguiar and Ferreira,

2013; Ferreira and Moreira, 1999). Despite their high costs, mechanical control measures are widely applied in Portugal (Moreira et al., 1999).

Although the growth and distribution of aquatic macrophytes in unshaded streams is mainly influenced by local hydraulic conditions (depth/velocity/sediments) (Chambers et al., 1991; Riis and Biggs, 2003), whose impact overshadows that of hydrochemistry (Steffen et al., 2014), little attention has thus far been paid to the possibility that channel encroachment and invasion can be controlled by establishing minimum annual flows. One common way of exploring the effectiveness of such ecosystem-regulation measures is ecological modelling, because model-based testing is faster and requires less financial inputs than actual physical experiments (Perona et al., 2009; Schmolke et al., 2010). Modelling species distribution or habitat suitability as functions of environmental factors is frequently used to provide spatial decision support for environmental management, weed or pest species risk assessments and studies of climate-change impacts (Franklin, 2013). In the case of river ecosystems, the instream flow incremental method (IFIM) (Bovee, 1982; Raleigh et al., 1986) is probably still the most widely used and accepted methodology for predicting the response of aquatic biota to the instream physical condition (Conallin et al., 2010; Jowett et al., 2008). However, its concepts have never been directly applied to the management of aquatic macrophytes.

Against this background, the main aim of this study was, for the first time, to apply and validate the hydraulic habitat modelling techniques promoted by the IFIM for the assessment of annual minimum flows with the ability to reduce the risk of channel encroachment and invasion by the alien *Myriophyllum aquaticum* in a heavily regulated Mediterranean river. Our hypothesis was that summer low flows further intensified by water abstraction for irrigation create physical instream conditions that favor the excessive growth of *Myriophyllum aquaticum* over the autochthonous *Sparganium erectum* and *Potamogeton crispus*, and that this situation can be mitigated by establishing minimum flows above a certain threshold.

2. Methodology

2.1. Study Area

The study area is located along the Sorraia river in central Portugal (Figure 35). The river basin has an accumulated area of 7719 km² and a semi-arid Mediterranean climate in which most of the annual rainfall (600–800 mm) occurs between October and May and the mean annual temperature is 16-19 °C. The fieldwork was carried out along a naturally braided, unconfined segment of the river. The riparian corridor from the edge of the active channel to the adjacent agricultural areas consists mostly of willow shrubs, and willows (*Salix alba*) in higher areas, and extends an average of 60 m either side of the river. The active channel has an average width of 15 m and is mostly unshaded. The segment's substrate is dominated by sands, gravels and cobbles. Surrounding land is given over to intensive rice, maize and tomato cultivation. We chose a calibration reach of approximately 1000 m in length for the model-building, and a model reach with a length of 320 m directly downstream for testing and application. Both reaches contain all the different mesohabitats (pool/run/riffle) found in the segment.
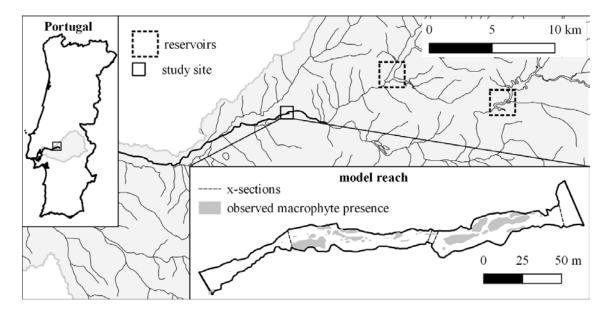
Figure 35. Location of the study site in Portugal and the Sorraia basin (rectangle), the position of the two largest reservoirs (dotted rectangles) and the wetted area of the model reach at Q = 0.3 m³/s, the location of the x-sections used for the hydraulic model calibration (including boundaries), and the observed macrophyte presence used to validate the habitat suitability model.

The Sorraia's hydrological regime presents a high intra- and inter-annual discharge variability, which is characteristic of Mediterranean watersheds (Gasith and Resh, 1999). The mean annual discharge is 20.14 m³/s (available data for 1933-1980, "Ponte Coruche" Gauging station). The heaviest winter floods can attain 887 m³/s, while during the summer months (June-September) the mean discharge is 3.2 m³/s and low flow spells are common (Figure 36).
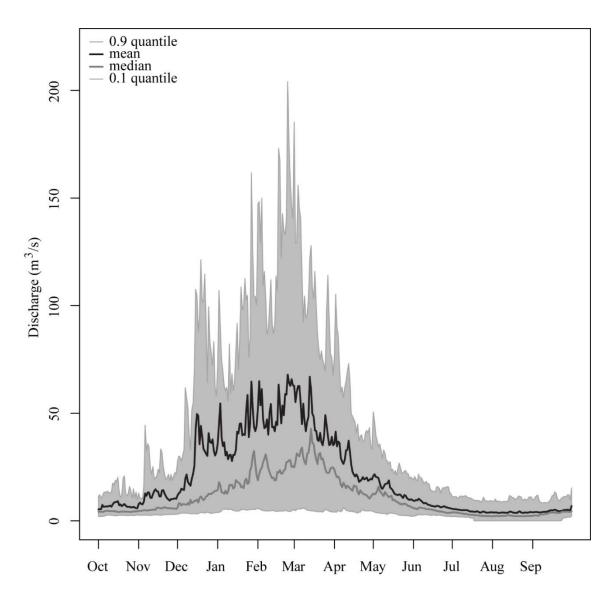
Figure 36. Summary of the flow regime of the Sorraia river (available data for 1933-1980 from the "Ponte Coruche" Gauging station): The area between the upper (0.9) and lower (0.1) quantiles is shaded grey; the black line represents the mean daily discharge; the grey line represents the median daily discharge.

The flow regime is heavily regulated by a system of reservoirs, weirs and canals that was implemented between 1933 and 1958. Water abstraction for agricultural irrigation

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is managed by a local farmers' association, which mechanically cleans the river channel of aquatic macrophytes and riparian vegetation every few years to reduce flood risk.

2.2. Aquatic vegetation

The main aquatic macrophyte species occurring in study area are *Myriophyllum aquaticum*, *Sparganium erectum* and *Potamogeton crispus*. Other species that presented less prevalence and were therefore not considered were *Ceratophyllum demersum* and *Typha domingensis.* Based on their growth form, *Myriophyllum aquaticum* and *Sparganium erectum* are classified as sediment-rooted plants with floating or emergent shoots/leaves, whereas *Potamogeton crispus* is a sediment-rooted submerged plant (Den Hartog and Van Der Velde, 1988). Following the definition of Pyšek et al. (2013) *Myriophyllum aquaticum* is considered an invasive species in Portugal. It was first reported in 1936 (Aguiar and Ferreira, 2013), but massive spreading was only observed in the 1970s (Moreira et al., 1999). *Myriophyllum aquaticum* is displacing native aquatic species, including *Potamogeton crispus* and *Ceratophyllum demersum*, in many parts of the River Tagus (Ferreira and Moreira, 1995).

2.3. IFIM overview

The instream flow incremental methodology (Bovee, 1982; Raleigh et al., 1986) is a framework which the U. S. Fish and Wildlife Services developed in the late 1970s to determine appropriate minimum annual flows by considering the effects of flow changes on instream habitat suitability of aquatic biota. It is probably still the most widely used and accepted methodology for predicting the response of aquatic biota to the instream physical condition (Conallin et al., 2010; Jowett et al., 2008). Its main feature is a hydraulic habitat suitability model that can be separated into a hydraulic component and a habitat component. The hydraulic model predicts water velocity, depth and other hydraulic variables. The habitat model is based on local habitat suitability curves (HSC) that describe the optimum range of a physical parameter affecting the species and are built on expert knowledge or field analyses of local

species occurrence and habitat availability. Integrating the two components makes it possible to calculate a composite suitability index (CSI) that combines the suitability information for each physical parameter at a given flow. The weighted usable area (WUA) for the target species is quantified by multiplying the composite suitability index by its area of influence. In order to assess an appropriate minimum annual flow, the hydraulic habitat suitability model is applied to a range of flows to produce a WUA-vs-discharge graph.

2.4. Hydraulic Habitat Suitability Modelling

In order to calibrate (train) the habitat suitability model, a total of 961 sample points were distributed systematically (2m x 2m), with a randomly chosen starting point along each mesohabitat (pool, run, riffle) found in the calibration reach. The mesohabitats were visually delimited in the field.

The occurrence of the main macrophyte species and physical habitat characteristics – flow velocity, water depth and grain size of the bed material – were analyzed at each sample point. The fieldwork was done in August 2016 and July 2017, during measured discharges of around 0.3 m³/s. We chose the lowest discharge period of the year in order to assess the hydraulic conditions during the period of least disturbance, which allows aquatic vegetation establishment and subsistence. Locations shaded by riparian vegetation (less than 5% of the analyzed reach) were excluded, since in this situation aquatic plant growth is mainly constrained by insufficient light (Carr et al., 1997). Depths were measured with a simple meter ruler and classified in intervals of 20 cm. Flow velocities were measured with a water flow probe (model FP101, Global Water Instrumentation, USA) positioned in the flow direction at 60 % of the flow depth and using 0.05 m/s intervals. The bed grain size was assessed visually and classified according to the Wentworth scale (sand: 0.62 - 2 mm; gravel: 2 - 64 mm; cobble: 64 - 256 mm). The habitat preferences for *Myriophyllum aquaticum*, *Sparganium erectum* and *Potamogeton crispus* were then calculated by dividing habitat-utilization (amount of species occurrences in each class of the physical parameters) by habitat-availability (total amount of each class of the physical parameters). The final preference values were normalized, from a minimum value of 0 for unsuitable to 1.0 for optimal habitats

(the class of the physical parameter with the highest amount of species occurrences), and expressed as a habitat suitability curve (HSC) for each physical parameter.

In order to apply and test the hydraulic habitat suitability model, we selected a 320 m-long reach directly downstream from the calibration reach. We chose a two-dimensional approach for the hydraulic simulation: the River2D model (Steffler and Blackburn, 2002). Two-dimensional hydraulic models predict depth and velocity laterally and longitudinally along the whole length of the river channel. They are therefore better able to simulate the complex flow patterns found in braided rivers than the more conventional (with regard to the IFIM) one-dimensional models that only predict depth and velocity across channel transects (Benjankar et al., 2015). The topography of the riverbed of the model reach, which is the main input into the hydraulic model, was measured in July 2016 with a Leica TCR703 Total Station (angle accuracy 3") along 970 points. The initial bed roughness values were estimated based on substrate size and vegetation distribution. To determine the boundary condition and calibrate the model, water depth and velocity were assessed along 6 transects including the down- and upstream cross-section, with measurements taken every 20 cm along the cross-section. The hydraulic model was calibrated by adjusting bed roughness until simulated water surface elevations matched measured water surface elevations.

The model was then used to simulate the physical instream conditions for a series of potential annual minimum flows of between 0.3 and 10 m³/s, representing a common flow range during the vegetation period. The weighted usable area (WUA) concept was used to evaluate the shift in habitat suitability for each discharge (Bovee, 1982). The WUA computation is based on the habitat suitability evaluated at every node of the topographic mesh and the "tributary area" of that node. We also calculated the Hydraulic Habit Suitability (HHS) for each discharge by dividing the WUA by the inundated area. The HHS can be understood as the percentage of the WUA from the inundated area at a given discharge. A value of 1 would mean that the whole of the wetted area classifies as usable area for a certain species or species group.

We used two different methods to calculate the habitat suitability. The classical, deterministic approach of the IFIM calculates a Composite Suitability Index (CSI) as the geometric mean of the separate suitability indices for depth, velocity, and substrate

size. It is directly integrated into the River2D Model on the basis of the HSC for each species.

$$CSI = \sqrt[3]{(VSI \times DSI \times SSI)}$$

VSI - Velocity Suitability Index; DSI – Depth Suitability Index; SSI – Substrate Suitability Index

In addition to the deterministic approach, we computed the habitat suitability for each species based on the random forest algorithm (RF) for classification (Breiman, 2001). We used the R package "randomForest" (Liaw and Wiener, 2002) to grow 1000 trees based on bootstrap samples of the same training data as that used to build the HSC, and incorporated 50% class weights into the classifier to account for the low prevalence of *Potamogeton crispus* and *Sparganium erectum*.

2.5. Model Validation

We mapped the true presence and absence of the main macrophyte species (*Myriophyllum aquaticum*, *Sparganium erectum* and *Potamogeton crispus*) in the model reach with a Global Positioning System unit (Ashtech, model Mobile Mapper 100; accuracy < 50 cm) during the same period (July/August) and with the same discharge (0.3 m³/s) as those when the data for the model calibration was collected. We then modelled the macrophyte distribution using the determenistic and the random forest approach based on the hydraulic simulation for the same discharge, and tested the agreement between observed and predicted distribution by assessing the area under the receiver operating characteristic curve (AUC) (Fielding and Bell, 1997). The AUC of a model is equivalent to the probability that the model will rank a randomly chosen species-presence site higher than a randomly chosen absence site. In addition, we transformed the predicted occurrence probabilities of both models to a binary presence/absence format for each species using the threshold of occurrence that maximizes the sum of sensitivity and specificity (Cantor et al., 1999; Liu et al., 2005). In order to assess the accuracy of the binary classification, we used the "True Skill Statistic" (TSS; sensitivity + specificity − 1), because it accounts for the effect of the

species prevalence (Allouche et al., 2006). All accuracy measurements were carried out using the R package "SDMtools" (VanDerWal et al., 2014).

In order to investigate whether our models accounted for all the factors causing the species' distributional pattern, we checked the observed species occurrence in the model reach for spatial autocorrelation using the Ripley's K function, and tested the error between observed and predicted species occurrence for clustering with the Moran's I index. The spatial analyses were done with the spatial statistics toolbox from ArcGIS for desktop (version 10.4.1)

3. Results

3.1. Habitat Suitability Curves

The habitat sampling resulted in 224 *Myriophyllum aquaticum*, 135 *Potamogeton crispus* and 85 *Sparganium erectum* presences in a total of 961 habitat samples.

Myriophyllum aquaticum displayed a substantial liking for low flow conditions, only having colonized areas with relatively slow velocities and low depth. It was already nearly absent at velocities over 0.1 m/s. The most suitable depths were 0-20 cm. In addition, it was found almost exclusively on sandy substrate. On the contrary, *Potamogeton crispus* seemed to prefer higher-flow areas. Its greatest presence occurred in medium velocities of 0.08 - 0.2 m/s and it clearly favored depths of more than 80 cm. Its preferred substrate was gravel. *Sparganium erectum* displayed a preference profile similar to that of *Myriophyllum aquaticum*, but was more tolerant of greater depth. The results show a distinct preference profile of the exotic *Myriophyllum aquaticum* with regard to flow velocity and water depth (Figure 37).
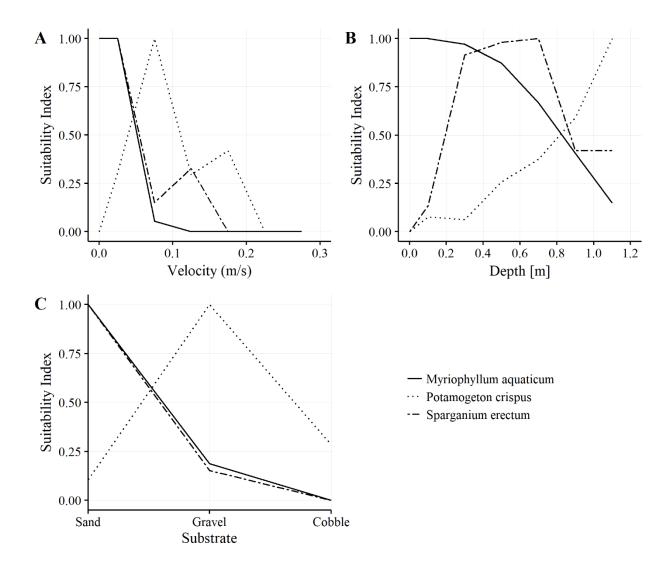
Figure 37. Suitability Index (SI) with regard to flow velocity (A), water depth (B), and substrate size of the bed material (C) for *Myriophyllum aquaticum*, *Potamogeton crispus* and *Sparganium erectum*; values of 1 signify optimal and values of 0 signify no suitability.

3.2. Model validation

In overall terms, the hydraulic habitat model based on the deterministic approach displayed a good discriminatory ability. In the case of *Myriophyllum aquaticum*, accuracy was even in the excellent range (AUC = 0.9), while for *Potamogeton crispus* it was good (AUC = 0.87), and for *Sparganium erectum* fair (AUC = 0.79). The performance of the binary classification differed more drastically between the species. Considering a threshold of occurrence for *Myriophyllum aquaticum* of 0.24, the TSS score of the model was 0.66. It correctly predicted 86% of the actual presences (sensitivity) and 80% of the actual absences (specificity). The occurrence

threshold for *Potamogeton* was set to 0.24. The TSS score was 0.62. Its occurrence was correctly predicted in 88% of cases, and its absence in 70%. The model's worst performance was for *Sparganium erectum*, with an occurrence threshold of 0.08 (TSS = 0.44; Sensitivity = 0.7; Specificity = 0.66).

The random forest model did not perform as well as the deterministic approach. On the contrary, only the prediction of *Myriophyllum aquaticum* achieved a similar accuracy (AUC = 0.85), whereas the predictions for *Potamogeton crispus* (AUC = 0.7) and *Sparganium erectum* (AUC = 0.65) were less accurate. This was also visible in the binary prediction. Considering a threshold of occurrence of 0.6 for *Myriophyllum aquaticum,* the model's TSS score was 0.66 (sensitivity = 0.8; specificity = 0.86). The prediction of *Potamogeton* based on a threshold of 0.5 returned a TSS score of 0.38 (sensitivity = 0.66; specificity = 0.72). Once again, the model performed worst for *Sparganium erectum* (threshold = 0.2; TSS = 0.28; sensitivity = 0.66; specificity = 0.62).

The species occurrence as well as the errors between the observed and predicted distributions presented a similar degree of positive spatial autocorrelation (clustered pattern), indicating that although our models have a medium to high degree of accuracy, they do not account for all the factors explaining species distribution.

3.3. Weighted Usable Area and Hydraulic Habitat Suitability

We only used the deterministic modelling approach to analyze the shifts in habitat suitability for incremental flows because of its better predictive performance.

The preference of *Myriophyllum aquaticum* for low flow conditions is also reflected in the development of the WUA. From 1167 m² at Q = 0.3 m³/s, it rapidly increases until it reaches its maximum of 3085 m² at Q = 1.4 m³/s. The WUA drops steadily after that, although the inundated and therefore potentially invadable area continues to increase with rising flows. The WUA decreases more slowly from Q = 5 m³/s to Q = 8 m³/s, after which it remains nearly constant. At Q = 0.3 m³/s *Potamogeton crispus* has a WUA of 1017 m², slightly lower than that of *Myriophyllum aquaticum* and *Sparganium erectum*. However, this then sharply increases, so that at Q = 3 m³/s the *Potamogeton crispus* WUA of 8004 m² is already 3 times higher than that of *Myriophyllum aquaticum*. After that, the upward trend continues more slowly, but steadily. At Q = 10 m³/s, the

Potamogeton crispus WUA of 10569 m² is over 10 times that of the invaders. The development of the WUA of *Sparganium erectum* initially appears to be similar to that of *Myriophyllum aquaticum*. However, it continues to gain area until Q = 3.5 m³/s, after which the WUA stays relatively constant at around 3900 m², whereas the *Myriophyllum aquaticum* WUA experiences a steady decline over the same range (Figure 38 – A).

In the case of *Myriophyllum aquaticum*, the HHS trends continuously downwards as discharge increases. Whereas 36% of the wetted area is potentially suitable at Q = 0.3 m³/s, only about 10% remains suitable at Q = 4 m³/s. *Potamogeton crispus* experiences an increase in HHS with rising flows. The HHS only decreases slightly at around Q = 1 m³/s, due to a large increase in wetted area. From Q = 3.5 m³/s onwards, the rate of change in HHS decreases. *Sparganium erectum* also experiences a decline in HHS, sharply at first, to levels below even those of *Myriophyllum aquaticum*, but remains nearly constant from Q = 2.5 m³/s onwards (Figure 38 – B).
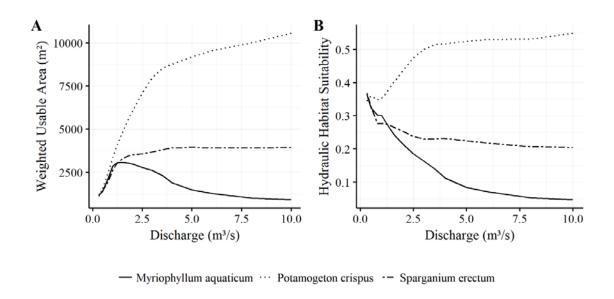
Figure 38. Weighted Usable Area (A) and Hydraulic Habitat Suitability (B) of the main species found in the study area as a function of discharge.

4. Discussion

In this study we wanted to explore setting minimum annual flows as an alternative management approach for controlling excessive growth of macrophytes and invasion

by *Myriophyllum aquaticum* during the vegetation period in the Sorraia river. Following IFIM principles, we built a hydraulic habitat suitability model for *Myriophyllum aquaticum*, *Sparganium erectum* and *Potamogeton crispus*, applied it to a range of discharges, and analyzed the changes in WUA and HHS. Our hypothesis was that low summer flows intensified by water abstraction for irrigation create physical instream conditions that stimulate excessive growth of *Myriophyllum aquaticum*, and that this situation can be mitigated by establishing minimum flows above a certain threshold.

The modelling results support our hypothesis that the growth and distribution of macrophytes in interflood periods is primarily a function of the local physical instream condition, which is especially favorable to an invasion of *Myriophyllum aquaticum* during the low flow range. Habitat suitable for *Myriophyllum aquaticum* already declines above flows of 1.4 m³/s, while the autochthonous species, and especially *Potamogeton crispus,* continue to gain ground. It would therefore seem possible to reduce the risk of invasion and favor a more natural species composition by setting annual minimum flows. The combination of the artificial approximation of the habitat availability for both the exotic and the autochthonous species caused by stable periods of flows under 1.4 m³/s and the greater competitive ability of *Myriophyllum aquaticum* may be the reason for the latter's successful expansion. Given that the mean annual flow during the vegetation period is 3.2 m³/s, it may well be that water managers can establish minimum annual flows above the 1.4 m³/s threshold and thereby avert this situation. This is an important result that can improve river restoration projects by preventing the degradation of natural aquatic vegetation communities.

However, we also observed that for the low flow range (0.3-1.4 m³/s), the WUA actually increases for *Myriophyllum aquaticum* and that the rate of change in habitat suitability for all species is lower with high flows than with low flows. The explanation for this is that the suitable areas are concentrated in shallow waters along the banks of the stream, and these shallow areas initially increase when the river enters the floodplain and then remain relatively constant in size. In the case of *Myriophyllum aquaticum*, this means that the WUA remains relatively constant above a discharge of 7 m³/s. Setting minimum annual flows will therefore not completely prevent an invasion; but it can contribute to an environmental flow regime that privileges autochthonous aquatic species and strengthens their competitive performance.

One major criticism of the IFIM habitat simulation to keep in mind when interpreting the results is the usage of the term Weighted Usable Area (Mathur et al., 1985), because it suggests a spatial extension of usable habitat when in fact it only actually describes the overall probability of use. So when we assess the effects of flow changes on aquatic biota, it is the shape of the WUA response curve that is more important than the magnitude (Jowett et al., 2008).

In addition, as with all modelling approaches, there are a number of different uncertainties that should be considered when interpreting the results.

4.1. Environmental factors

Our study is based on the assumption that in hydrologically stable periods, physical habitat characteristics are the main limiting factor for aquatic species in streams. Indeed, several studies argue that flow velocity is the main environmental factor controlling the abundance and distribution of aquatic macrophytes (Baattrup-Pedersen and Riis, 1999; Chambers et al., 1991; Janauer et al., 2010; Madsen et al., 2001). Most studies relate the limiting effect of higher flow velocities on plant growth to increased drag forces on the plants and their anchoring ground, causing uprooting, or less frequently, stem breakage (Chambers et al., 1991; Riis and Biggs, 2003). However, a more recent study (Pollen-Bankhead et al., 2011) indicates that the preference of macrophytes for low velocities is less related to the drag forces on the plants and more to the conditions controlling erosion and deposition of fine substrate materials. The effect of substrate size has mainly been studied with regard to the distribution patterns of macrophytes, and not in terms of changes in biomass (Baattrup-Pedersen and Riis, 1999; O'Hare et al., 2006; Riis and Biggs, 2001). The findings indicate a niche separation between macrophytes based on different substrate size preferences. Apparently, submerged species favor coarser substrates (gravel and boulder), whereas species that grow both submerged and emergent, and species that only grow emergent, were associated with finer substrates (sand) typical of low flow conditions. This is coherent with our results. The influence of flow depth has been related to light availability, which decreases with greater depth (Koch, 2001). In situations of high turbidity or direct shading, for example through overhanging vegetation, light availability can also become the main limiting factor, which is why we excluded sample

sites with these characteristics (Köhler et al., 2010). Temperature is also known to influence the growth rate of aquatic plants (Koch, 2001). It can, however, be assumed that temperature alterations in the analyzed flow range are marginal and are indirectly covered by the effects of velocity and depth (Gu et al., 1998). Besides the physical factors, geochemical properties of the stream and especially nutrient availability are known to have an influence on aquatic biota (Koch, 2001). Unnatural high concentrations of phosphorus, as often occur in agricultural watersheds, can stimulate excessive macrophyte growth (Mainstone and Parr, 2002). However, these factors are still most probably overshadowed by the hydraulic conditions (Barendregt and Bio, 2003; Steffen et al., 2014), as is also indicated by the high accuracy of our model.

4.2. Data collection / Model calibration

Different forms of data analysis for generating the HSC for each environmental factor are distinguished for the IFIM (Bovee, 1986): a) expert knowledge; b) analyses of actual habitat conditions used by the species (or presence only data); and c) in-situ species occurrence and habitat availability data (or presence/absence data). We based our model calibration solely on actual presence/absence data (c). It is the most highly recommended of the three methods (Jowett et al., 2008), and the only one that permits an estimation of the true probability of observing a species at a site (Guillera-Arroita et al., 2015). We kept geographical sampling bias to a minimum by selecting a calibration (training) reach and a model reach from the same river segment, and by applying a stratified, systematic sampling design with a random starting point. The detection error, which is crucial to the performance of many habitat suitability models (Lahoz-Monfort et al., 2014), can be considered negligible because of the sampling design, the small number of different species and their sessility.

Model calibration errors can also affect the two-dimensional hydraulic modelling, which can be compromised due to the collection of insufficient or erroneous bed topography data, insufficiently detailed substrate distribution mapping, erroneous model calibration, or failure to include effects of the bed topography upstream of the study site in the model (Jowett and Duncan, 2012).

4.3. Model algorithm

The IFIM commonly uses a univariate algorithm to relate the abiotic characteristics to actual habitat suitability (Conallin et al., 2010). The univariate derivation of the composite suitability index is criticized for being based on the assumption that organisms select each habitat variable independently, ignoring interactions and cumulative effects between them (Ahmadi-Nedushan et al., 2006), such as the influence of velocity on substrate stability and composition (Shields, 1936). Multivariate statistical models, such as Generalized Additive Models (Milner et al., 2001) and Artificial Neural Networks (Gozlan et al., 1999), are alternative means of fitting the suitability data that are able to account for interactions between the variables and overcome the problem of independence (Ahmadi-Nedushan et al., 2006). Another, increasingly popular, approach is the use of "fuzzy logic" to define a set of rules that classifies suitability according to a combination of different environmental factors. It allows consideration of uncertain measurements and vague expert knowledge, as well as multivariate effects, without requiring the input parameters to be independent (Noack et al., 2013). With random forests we also applied a distribution modelling technique that is capable of modelling complex interactions among predictor variables and is considered to have one of the greatest discriminatory capacities (Cutler et al., 2007; Elith et al., 2006).

However, random forest and all other approaches are static and ignore more complex processes that are known to shape the distribution patterns of macrophytes, such as interspecific competition and feedbacks between the plants and the physical environment known as niche construction (Corenblit et al., 2009). The latter has become very evident in the complex relationship between macrophytes and fine sediment, where macrophytes have been observed to create positive growth conditions through retention and stabilization of fine sediments, thereby also interacting with geomorphological processes (Schoelynck et al., 2012).

4.4. Model validation

Ecological modelling is of little value if the prediction is not tested against independent data (Olden et al., 2002). We therefore separated the study reach from the calibration reach and collected field data in two different years. The overall model prediction

capacity at Q = 0.3 m³/s was assessed as good using the threshold-independent AUC statistic. The binary prediction, and especially the rate of observed absences of the species that fall in pixels of predicted presences (the commission error rate, which equals 1 minus specificity), was less convincing, but can in part be explained by the low prevalence of the species. A distinction must be made between two different types of commission error: real commission errors, in which combinations of environmental conditions that are not within the species' niche are falsely interpreted as suitable; and apparent commission errors, where absence represents a real feature of the species' distributional ecology due to interspecific interactions and historical factors (Peterson, 1999). A high commission error is therefore common among species that show a low prevalence, and can be an indicator that the species has not yet conquered the whole of its potential niche. If this interpretation is correct, it would support the use of our model as a screening tool for identifying areas that are at higher risk of invasion.

We can only speculate about the causes of the spatial autocorrelation in the errors between observed and predicted species distribution: disregard of interactions between the predictor variables, omission of important predictors (temperature, nutrients), or ecological processes (dispersal, competition, niche construction) (Guisan and Thuiller, 2005). However, the model's good predictive performance against independent data nonetheless proves the usefulness of the IFIM approach for predicting macrophyte distribution.

4.5. Other management options and conclusion

Mechanical methods are the most widely used measures for controlling aquatic macrophytes in both Portugal (Moreira et al., 1999) and Europe as a whole (Hussner et al., 2017). They allow for containment or eradication, depending on the specific technique and frequency of application (Madsen, 2000). Although often regarded as environmentally less harmful, the most common and effective measures like mowing are not species-specific and can both harm non-target aquatic biota and cause sediment resuspension (e.g. Habib and Yousuf 2014). Worldwide, chemical control is also applied. While proven very effective, even for eradicating nuisance weeds (Champion and Wells, 2014), herbicides will physiologically affect similar native aquatic plants and potentially also indirectly harm fish and invertebrates (Getsinger,

1998). The use of herbicides to control aquatic nuisance weeds is therefore severely restricted in various countries (especially in the EU). Biological measures also present a risk of off-target impacts, both directly and indirectly through alteration of the food web. Physical management methods are distinguished from mechanical techniques, because instead of the plants directly, it is their environment that is manipulated. Several physical techniques can be distinguished: dredging, drawdown, benthic barriers, shading or light attenuation, and nutrient inactivation (Madsen, 2000; Wersal et al., 2013). The control of nuisance weeds through flow regulation fits into the latter category, but has so far received little attention. Flushing flows have been successfully used to eradicate weeds in the Ebro river (Tena et al., 2013). However, frequency and magnitude of discharges (in the range of a 2-year flood) are not a viable option for intensive agricultural watersheds like the Sorraia, where both the side effects of the floodings and the competing water uses have to be considered.

Although most management techniques have some negative side effects on the ecosystem, so do the invasion and extreme growth of alien species. Maintaining minimum discharges in order to prevent channel encroachment may be an ecologically and financially advantageous addition to the range of commonly practiced control measures. We tested this approach by applying habitat suitability modelling techniques that are widely used to evaluate environmental flows and restoration measures aimed at fishes and invertebrates. Based on the specific habitat preferences of *Myriophyllum aquaticum*, it seems possible to set minimum flows that reduce the invader's habitat while simultaneously promoting that of autochthonous and less invasive aquatic species. This measure can be recommended with a high level of confidence, given that when the model was checked against independent data, it displayed a good level of accuracy in predicting species distribution.

Acknowledgments

SECTION V

ECOLOGICAL FEEDBACKS OF RIPARIAN VEGETATION MANAGEMENT

CHAPTER 7

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

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Resumo

Os caudais ambientais permanecem enviesados no sentido do tradicional grupo biológico da ictiofauna. Consequentemente, estes caudais ignoram a variabilidade interanual do regime de escoamento que governa as espécies de ciclos de vida mais longos, negligenciando por isso a perspetiva de longo prazo do ecossistema fluvial. Analisou-se a importância de considerar os requisitos ripários na eficiência de longo prazo dos caudais ambientais. Para essa análise, modelou-se o desenvolvimento da vegetação ripária durante uma década em função de diferentes caudais ambientais em dois locais de estudo. Posteriormente, avaliou-se a disponibilidade de habitat piscícola correspondente de três habituais espécies piscícolas para cada um dos resultantes cenários de paisagem ripária. Os resultados da modelação demonstraram que os caudais ambientais desconsiderando os requisitos de vegetação ripária permitiram a degradação ripária, particularmente a invasão do canal fluvial. Tal circunstância alterou as características hidráulicas do canal fluvial onde as alturas e velocidades de escoamento sofrem alterações locais de até 10 cm e 40 cm s^{-1}, respetivamente. Consequentemente, após uma década deste regime de caudais, a área de habitat disponível para as espécies piscícolas consideradas experiencia modificações até 110% quando comparada com o habitat natural. Por sua vez, os caudais ambientais que consideram os requisitos de vegetação ripária foram capazes de manter a vegetação ripária perto dos padrões naturais, preservando desta forma as características hidráulicas do canal fluvial e sustentando o habitat piscícola numa condição aproximadamente natural. Consequentemente, a disponibilidade de habitat piscícola nunca se alterou mais de 17% do habitat natural.

Palavras-chave: vegetação ripária, fauna piscícola, microhabitat aquático, caudais ambientais, regularização fluvial, requisitos de caudal

Abstract

Environmental flows remain biased toward the traditional biological group of fish species. Consequently, these flows ignore the inter-annual flow variability that rules species with longer life cycles and therefore disregard the long-term perspective of the riverine ecosystem. We analyzed the importance 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 landscape 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 local 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 up to 110% when 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 17% from the natural habitat.

Keywords: riparian vegetation, fish fauna, aquatic microhabitat, environmental flows, river regulation, flow requirements

IMPORTANCE OF CONSIDERING RIPARIAN VEGETATION REQUIREMENTS FOR THE LONG-TERM EFFICIENCY OF ENVIRONMENTAL FLOWS ON AQUATIC MICRO-HABITAT

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Author contributions

Conceived and designed the experiments: RR IB JMS. Performed the experiments: RR IB JMS. Analyzed the data: RR IB JMS AP. Contributed reagents/materials/analysis tools: RR IB. Wrote the paper: RR IB JMS AP TF.

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1. Introduction

Freshwater ecosystems provide vital services for human existence but are on top of the world's most threatened ecosystems (Revenga et al., 2000; Dudgeon et al., 2006), 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). Actually, 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., Maheshwari et al., 1995; Nilsson and Berggren, 2000; FitzHugh and Vogel, 2010; Miller et al., 2013; Uddin et al., 2014b; a). As a result of this, 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 play an essential role in the conservation of freshwater ecosystems (Hughes and Rood, 2003; Arthington et al., 2006). It is now in agreement that environmental flows must ideally be based on the ecological requirements of different biological communities (e.g., Poff et al., 1997; Arthington and Zalucki, 1998; Davis and Hirji, 2003; Dyson et al., 2003; Acreman et al., 2009; Acreman and Ferguson, 2010; Arthington et al., 2010; Arthington, 2012; Acreman et al., 2014) 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; Postel and Richter, 2003; Lytle and Poff, 2004). In this sense, holistic methodologies meant to address river systems as a whole (Arthington et al., 1992; King and Tharme, 1994; King and Louw, 1998) are clearly being increasingly applied out of Australia and South Africa (Hirji and Davis, 2009), the origin countries of this holistic concept. However, 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 (Tharme, 2003; Acreman et al., 2009; Arthington, 2012), and require an input from less typically monitored taxa (Gillespie et al., 2014). Accordingly, these approaches still disregard the inter-annual flow variability that rules species with longer lifecycles, like riparian vegetation, therefore lacking the long-term perspective of the riverine ecosystem (Stromberg et al., 2010b). 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 vegetation is a suitable environmental change indicator (Nilsson and Berggren, 2000; Benjankar et al., 2012) that responds directly to flow regime in an inter-annual timeframe (Poff et al., 1997; Naiman et al., 2005; Capon and Dowe, 2007) and has a clear significance in the habitat improvement of aquatic systems (e.g., Gregory et al., 1991; Tabacchi et al., 2000; Pusey and Arthington, 2003; Broadmeadow and Nisbet, 2004; Dosskey et al., 2010; Salemi et al., 2012; Statzner, 2012; Wootton, 2012; Ryan et al., 2013; Van Looy et al., 2013; Rood et al., 2015; Chase et al., 2016). In fact, riparian vegetation and aquatic species interact biologically, physically and chemically (Gregory et al., 1991). Riparian vegetation is capable of influencing aquatic species in several ways. It affects food webs by providing an important input of nutrients that are a major food source for invertebrates, which are in turn eaten by fishes (Wootton, 2012). It influences hydrological processes (Tabacchi et al., 2000; Salemi et al., 2012) and protects aquatic habitats by means of river bank stability (Rood et al., 2015) and providence of large woody debris (Fetherston et al., 1995). It provides thermal regulation of rivers by overshadowing (Ryan et al., 2013) and protect water quality both by trapping sediments and contaminants (Chase et al., 2016) as by chemical uptake and cycling (Dosskey et al., 2010). On the other hand, aquatic species appear also to be able to influence riparian zones, although in a much smaller magnitude, acting as ecosystem engineers (Statzner, 2012). For instance, fishes can dig in sand and gravel for food or reproductive purposes and therefore influence sediment surface characteristics and critical shear stress (e.g., Statzner et al., 2003; Hassan et al., 2008).

Accordingly, 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). 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.

We have already noticed how environmental flow regimes disregarding riparian vegetation requirements allow for the degradation of riparian woodlands in the subsequent years following such river regulation (e.g., Rivaes et al., 2015). However, we are not aware of studies assessing the return effect of this degradation again on the efficiency of those environmental flow regimes. 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 landscape degradation on fish species. By riparian landscape we mean the specific spatial patterns of riparian vegetation that result from ecological, geomorphological and hydrological processes, and are depicted by the existing patch mosaic with different vegetation types and succession phases. We were particularly interested in answering the following questions: i) are environmental flows exclusively addressing fish requirements capable of preserving the habitat availability of these aquatic species in the long-term? ii) If not, to what extent can the disregard for riparian vegetation requirements derail the goals of environmental flows addressing only aquatic species as a result of the riparian landscape degradation? iii) 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 (please see Naiman et al., 2005, and NRC, 2002 for a better understanding about riparian vegetation structure) 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 landscape 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 39). 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 (Gasith and Resh, 1999), with a low flow period interrupted by flash floods in winter (median of mean daily discharges in the winter months is 8.8 m^3 s^{-1} and maximum annual discharges with a return period of 2, 5, 10 and 100 years are respectively 323, 549, 718 and 1314 m^3 s^{-1}) and a very low flow, even null at times, during summer (the first quartile and median of mean daily discharges in summer months is respectively 0 and 0.1 m^3 s^{-1}). 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 located 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).
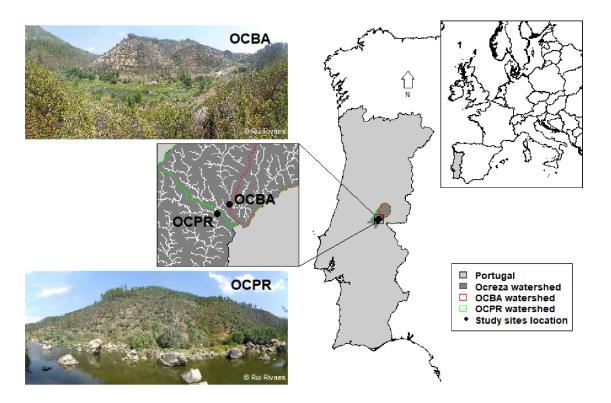
Figure 39. Location and characterization of the study sites OCBA and OCPR.

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 (Appendix E – Tables E1, E2, E3 and E4).

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 100), while dendrochronological methods were used to determine the approximate age of the patches. Two or three of the largest individuals in each patch were cored with a standard 5 mm increment borer, taking two perpendicular cores at breast height in adult trees (Mäkinen and Vanninen, 1999). For individuals with a diameter smaller than 5 cm at breast height, discs were obtained for age calculation purposes, and on multistemmed trees, the cores/discs were taken from the largest stem. 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 in 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 (Appendix E – Table E5, Table E6, Figures E1 and Figure E2).

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; Boavida et al., 2015). Fish 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 five seconds. Upon sighting a fish or a shoal of fishes, a numbered location marker was anchored to the streambed 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. Ensuing fish sampling, microhabitat measurements of flow depth (cm), mean water velocity (cm s^{-1}) and dominant substrate composition were taken in 0.8 by 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 data, as well as data

analysis to determine habitat use, availability and preference of fish species regarding the considered variables, is provided as supplementary material (Appendix E – Table E7 and Figure E3 to Figure E12).

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 natural flow regime data was obtained from the Portuguese Water Resources National Information System (SNIRH, 2010). The environmental flow regimes used in this study are an adaptation from the environmental flow regime created by Ferreira et al. (2014) for the location of the study sites (Figure 40). These authors determined an environmental flow regime presented in a multiannual fashion considering a decadal time frame and accounting for two different flow regime components: a monthly flow regime addressing fish requirements and a multiannual flow regime composed by floods with different recurrence intervals addressing riparian vegetation requirements. The first component, i.e., the flow regime addressing fish requirements (Eflow), was determined according to the Instream Flow Incremental Methodology (Bovee, 1982) and was built on a monthly basis to embody the intra-annual variability ruling the main life cycle events of this biological group (Gasith and Resh, 1999; Encina et al., 2006). These mean monthly discharges addressing fish requirements that compose the Eflow 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 second component of the environmental flow regime (floods with a certain recurrence interval) proposed by Ferreira et al. (2014) was determined according to Rivaes et al. (2015) and intend to characterize the inter-annual flow variability to which the arrangement of riparian vegetation communities respond (Hughes, 1997). The flushing flows addressing

riparian requirements in the Eflow&Flush regime 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). Therefore, the environmental flow regimes used in this study are considered an adaptation from Ferreira et al. (2014) as we used just the fish-addressing component (only mean monthly discharges) as the standard procedure of an environmental flow regime considering only fish requirements (Eflow) and both components (mean monthly discharges and flushing flows) for the environmental flow regime addressing fish and riparian requirements (Eflow&Flush).
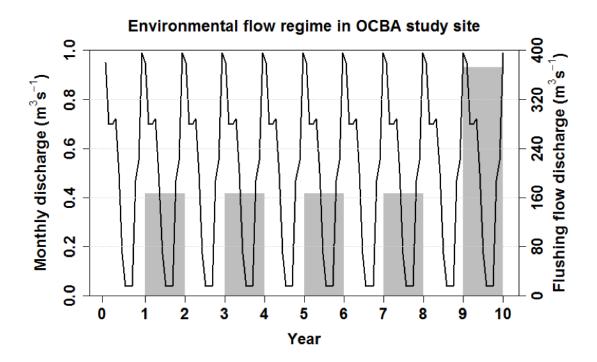
Figure 40. Environmental flow regime addressing fish (black line, left axis) and riparian (grey bars, right axis) requirements considered for the habitat modeling in OCBA study site. Fish requirements are addressed by a constant monthly discharge and riparian requirements by a flushing flow in the years in which are planned (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.

2.4. Riparian vegetation modeling

The riparian vegetation modeling was performed using the *CASiMiR-vegetation* model (Benjankar, 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 strengths of this model are the capacity of incorporating the past patch dynamics into every model run, the ability of working at a response guild level by using succession phases as modeling units, and the ability of providing the outputs in a spatially-explicit way. In turn, main disadvantages of this model can be attributed to the inexistence of a plant competition module or the lack of an incorporated hydrodynamic model.

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 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. The groundwater depth corresponding to the mean annual discharge of the river is also a model input used as a reference for the general habitat conditions that determine the expected riparian landscape according to the calibrated thresholds of the riparian succession phases. Thus, the magnitude and duration of extreme low flows are accounted by CASiMiR-vegetation model. A complete detailing of model rational and parameterization can be found in Politti and Egger (2011) and Benjankar et al. (2011). Model calibration was carried out in accordance with the methodology described in previous studies (García-Arias et al., 2013; Rivaes et al., 2013). Particularly, calibration was performed by running the CASiMiR-vegetation model for a decade to simulate the effect of the local historic flow regime on riparian vegetation. The result of the model was then compared with an observed vegetation map that was surveyed in the same year of the one corresponding to the result of the model. This is an iterative process of trial an error where the parameter of shear stress resistance threshold of each succession phase is tuned to obtain the best calibration outcome (see Wainwright and Mulligan, 2004, for a better understanding). All the other parameters, namely, patch

age and height above water table ranges were determined based on the data collected in the field. This information is provided as supplementary material (Appendix E – Table E5). 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). This agreement evaluation can be understood as a classification 61% better than what would be expected by a random assignment of classes. The riparian vegetation model was further validated in this specific watershed (Ferreira et al., 2014) with even better results (quadratic weighted kappa of 0.68). After calibration and validation (calibrated parameters provided as supplementary material; Appendix E – Table E5), the riparian vegetation was modeled for periods of ten years according to the corresponding flow regimes (Table 10). 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). Furthermore, during modeling, riverbed topography was considered fixed for several reasons: the study sites are located in a fairly steep valley in which river is not allowed to meander considerably during such a short time scale; the typical substrate of both study sites is armored and very coarse (boulders, large boulders and bedrock); in these conditions the small monthly discharges intended to maintain aquatic fauna requirements are not able to create water depths and flow velocities capable of moving or eroding particles with the size of those found as substrate in the considered study sites (for a better understanding please see (Hjulström, 1939; Clarke and Hansen, 1996; Alexander and Cooker, 2016); no significant differences were found during the substrate analysis of the different succession phases; prior knowledge of the authors show that the considered floods do not bring noteworthy changes to river geomorphology during this period (Rivaes et al., 2015); the model calibration and validation results exhibited a good agreement with the observed riparian landscape while using the same methodology; by using a fixed topography it is possible to analyze the exclusive effect of riparian landscape degradation on the river hydraulics.

The resulting riparian vegetation maps were then used as the respective riparian landscapes (hereafter named natural, Eflow and Eflow&Flush landscapes) in the hydrodynamic modeling of fish habitat in each study site.

Table 10. Maximum annual discharges ($m^3 s^{-1}$) considered in the CASiMiR-vegetation model for each study site.

Year	OCBA			OCPR		
	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

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 (Jalón and Gortázar, 2007; Boavida et al., 2011) 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 strengths of this model are the fact of being public domain software and to be technically robust throughout a wide range of modeling circumstances. On the other hand, some limitations of this model are the non-incorporation of a morphodynamic module or the ability of embodying fuzzy logic rules during the computation of species habitat availability.

The calibration procedure followed the methodology proposed by (Boavida et al., 2013a; Boavida et al., 2015). Calibration was performed by iteratively adjusting the bed channel roughness to attain a good agreement of the simulated versus surveyed water surface elevations and velocity profiles in the surveyed cross-sections. Boundary conditions were set according to the water surface elevations measured at the upstream and downstream cross-sections. Calibrated parameters are provided in supplementary material (Appendix E – Tables E1, E2, E3 and E4).

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 landscape scenario. The different riparian landscapes were represented in the hydrodynamic model by changing the channel roughness according to the spatial extent of the riparian succession phases, i.e., the channel roughness inputted to the model are the riparian landscape maps converted into channel roughness maps. Roughness is a critical feature influencing the physical variables of flow hydraulics (Chow, 1959; Curran and Hession, 2013), 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.

After modeling the Eflow discharges in each of the riparian landscape scenarios of the two study sites, the hydraulic characteristics of each riparian landscape (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 riparian landscapes. 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 (note that this *aquatic zone* is a definition *sensu* CASiMiR-vegetation model, designating the area of the river channel that is permanently submerged throughout the hydrologic year and where riparian vegetation is unable to establish and develop. It corresponds to only a fraction of the wetted area by river flow during the discharges considered in the subsequent hydrodynamic modeling) 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):

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

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):

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

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.

2.6. Workflow of the modeling procedure

The workflow of the modeling procedure is presented in Figure 41. Firstly, the calibrated version of the riparian vegetation model is used to produce the riparian landscape scenarios according to each of the considered flow regimes. In each modeling run, this model uses as inputs one of the specific flow regimes mentioned and models the effects of a decade of such flow regime in the local riparian vegetation.

The output of the model is an expected riparian vegetation landscape map (detailed by succession phases) resulting from the inputted flow regime. This map is converted into a channel roughness map by attributing to each riparian succession phase a specific effective roughness height based on the expert knowledge of the authors, on literature (e.g., (Chow, 1959; Barnes, 1967; Fisher and Dawson, 2003) and on the calibration results of the models. The considered roughness values of each succession phase are provided as supplementary material (Appendix E – Tables E3 and E4). These roughness maps are one of the inputs of the River2D model.
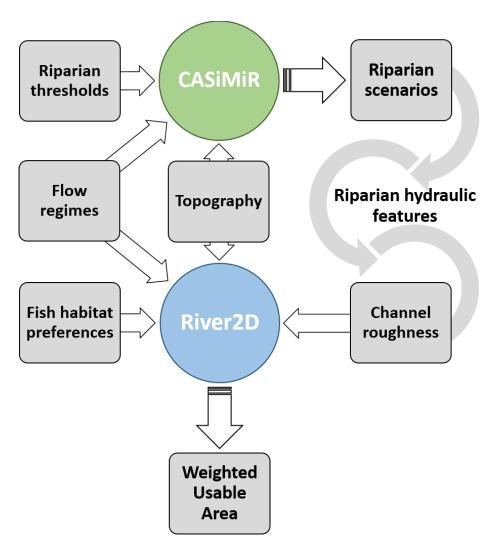
Figure 41. Methodological scheme representing the workflow of the modeling procedure. White arrows stand for direct inputs, striped white arrows for model outputs and grey arrows for variable conversion processes.

Secondly, the hydrodynamic model River2D is used to determine the water depths and flow velocities at the microhabitat scale (already considering each of the roughness maps coming from the conversion of the CASiMiR-vegetation output vegetation maps) and to compute the weighted usable areas of the considered fish species using the previous calculated variables and the inputted information regarding the observed fish species habitat preferences for water depth and flow velocity. This is done similarly using every of the riparian landscape scenarios. For each scenario run, the outcome of this model is therefore the weighted usable area of each of the considered species and life stages for each of the discharges considered in the Eflow regime.

3. Results

3.1. Riparian vegetation modeling

Different riparian landscapes resulted from the riparian vegetation modeling according to the considered flow regimes in both case studies (Figure 42). Nonetheless, the modeled response of riparian vegetation to each flow regime is similar in the two study sites. The riparian landscape, 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 riparian landscape, 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 landscape 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.
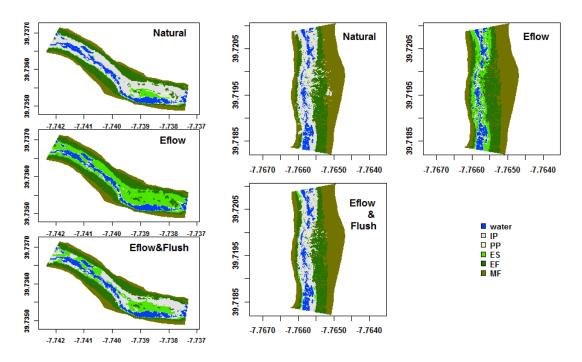
Figure 42. Expected patch mosaic of the riparian vegetation habitats shaped by the natural, Eflow and Eflow&Flush flow regimes (detailed by succession phase, namely, initial phase – IP, pioneer phase – PP, early succession woodland phase – ES, established forest phase – EF and mature forest phase – MF) in the OCBA study site (on the left) and in the OCPR study site (on the right).

The riparian landscape driven by the Eflow&Flush regime shows the capacity of this flow regime in hold back vegetation encroachment in both cases. In this riparian landscape scenario, 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.

Summing up, the results of the riparian vegetation modeling show a riparian landscape degradation by vegetation encroachment in the Eflow landscape scenario when compared with the natural riparian landscape. Instead, the Eflow&Flush landscape scenario keeps approximately the same patch disposal and succession phase's proportion as the natural landscape and therefore does not present evidence of riparian landscape degradation.

3.2. Hydrodynamic modeling

The changes undertaken by the riparian vegetation facing different flow regimes are able to modify the hydraulic characteristics of the river stretches (Figure 43). Channel

effective roughness heights (k_s) change dramatically according to the considered riparian landscapes, increasing proportionally to the encroachment level of vegetation in the study sites. In both case studies, the k_s values of the Eflow landscape are clearly distinct and higher compared to the other two riparian landscapes (Figure 43). The k_s values in the Eflow&Flush landscape were found to be between the values of Eflow and natural landscapes in the case of OCBA, and were very similar with the natural landscape in the case of OCPR (Figure 43). Notwithstanding, in both case studies, the k_s mean values are statistical significantly different between all three riparian landscapes (test results in supplementary material; Appendix E – Table E8). The mean k_s of the Eflow, Eflow&Flush and natural landscapes 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 (Figure 43). Although not so noticeable due to the great amount of data, differences are statistically significant. In OCBA, the Eflow landscape creates a circumstance with statistically significant 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 landscapes. The t-tests on water depths (H0: true difference in means is equal to 0) revealed highly significant p-values (<0.001), respectively, for the comparisons between Eflow and natural flow regimes, and Eflow and Eflow&Flush flow regimes. The t-tests on flow velocities also derived a highly significant p-value (<0.001) in both the comparisons of natural versus Eflow regimes and Eflow versus Eflow&Flush flow regimes (test results in supplementary material; Appendix C – Tables C2 and C3). In contrast, depth and flow velocity are not significantly distinguishable between the natural and Eflow&Flush landscapes, 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 (t-tests obtained p-values of 0.122 for natural versus Eflow regimes and 0.098 for Eflow versus Eflow&Flush flow regimes). Mean values of flow depth for Eflow, Eflow&Flush and natural landscapes are 0.420, 0.417, 0.418, respectively. Nonetheless flow velocities are different with statistical significance as the p-values of the t-tests for natural versus Eflow and for Eflow versus Eflow&Flush were highly significant (<0.001). The Eflow landscape creates statistical significantly lower flow velocities (0.271 m s^{-1}) when

compared to the statistical significantly indistinct Eflow&Flush (0.277 m s^{-1}) and natural (0.278 m s^{-1}) landscapes (test results in supplementary material; Appendix E – Table E9 and Table E10).
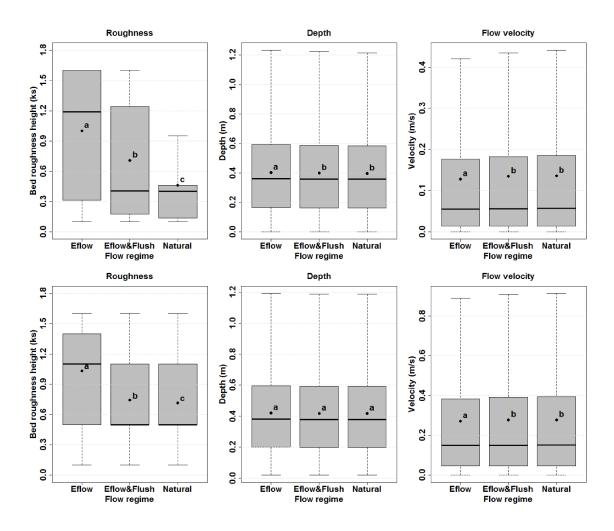
Figure 43. Hydraulic characterization of OCBA (top) and OCPR (bottom) according to the different expected riparian vegetation habitats driven by the Eflow, Eflow&Flush and natural flow regimes (data obtained from 2D hydrodynamic modeling). Different letters stand for statistical significant differences between groups (t-test). Boxplots portray non-outlier value range, thick black lines the median value and black dots the mean values.

Furthermore, when comparing water depths and flow velocities point by point, one can find differences between scenarios up to 10 cm in water depth and more than 40 cm s^{-1} in flow velocity. Accordingly, there are locations where the considered hydraulic parameters change considerably, shifting the habitat preference of fishes in one or two classes of the corresponding habitat preference curves.

In general, the Eflow landscapes present an increased channel roughness interfering with river flow and creating increased water depths and slower flow velocities when compared with the natural landscape. On the contrary, despite the increased channel roughness of the Eflow&Flush landscape, the water depths and flow velocities are very similar to the ones in the natural landscape. These results demonstrate that an environmental flow addressing exclusively fish requirements is not capable of preserving the habitat availability of the aquatic species for which was proposed in the long-term.

3.3.   Analysis of the aquatic habitat suitability for fish species

During a hydrological year, each riparian landscape provides different WUAs for the target fish species, with the same environmental flow regime addressing fish species (Figure 44). Differences from the natural habitat suitability are greater in the Eflow landscape 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 landscape, the WUA modifications instilled by the Eflow landscape are on average approximately 12%, and are higher than 17% in a quarter of the cases reaching 80% in an extreme situation. Particularly, the Eflow landscape provides less habitat suitability during autumn and winter months for the barbel and nase juveniles, c. 17% and 14%, respectively. Likewise, in this riparian landscape, 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 landscape provides a WUA very similar to the natural landscape. The habitat changes created by the Eflow&Flush landscape 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 landscape 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 landscape consistently provides less habitat suitability during autumn and winter

months for the nase juveniles and adults, c. 50% and 38%, respectively, while the habitat suitability increases in approximately 46% of calandino. Moreover, the Eflow landscape 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 landscape provides a WUA very similar to the natural landscape throughout the year. The habitat changes created by the Eflow&Flush landscape 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 landscape revealed to be significant in several months by the χ^2 test whereas this were never the case for the Eflow&Flush landscapes (test results provided in supplementary material; Appendix E – Table E11, Table E12, Table E13 and Table E14).

The riparian-induced modifications on the WUAs are also confirmed by all the employed deviation measures (Table 11). According to RMSD, MAD and MAPD, the habitat provided by the Eflow landscape 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. All together, these results reveal that the overlook of riparian requirements into environmental flows can derail the goals of environmental flows addressing only aquatic species by an extent of approximately an average of 12 to 29% of the fish WUA's in the considered study sites as a result of the riparian landscape degradation. On the other hand, results reveal that environmental flows regarding riparian requirements are able to maintain the habitat availability of fish species as the WUA's in the study sites never change on average more the 3% in a decade.
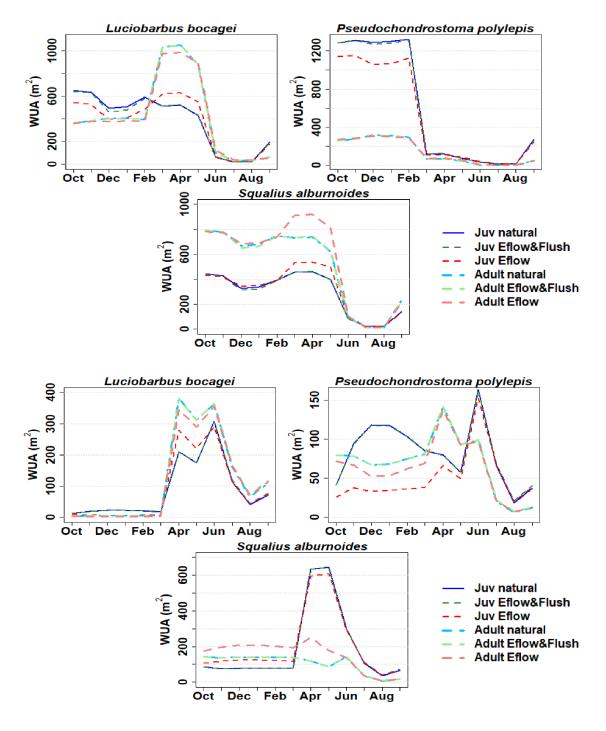
Figure 44. Fish weighted usable areas provided by the fish-addressed environmental flow regime (Eflow) flowing through the different riparian landscape scenarios originated by a decade of three different flow regimes (natural, Eflow&Flush and Eflow) at the OCBA (top three graphics) and OCPR (bottom three graphics) study sites.

Table 11. 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 (m²)	MAD (m²)	MAPD (%)	RMSD (m²)	MAD (m²)	MAPD (%)	RMSD (m²)	MAD (m²)	MAPD (%)	RMSD (m²)	MAD (m²)	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

4. Discussion

This study evaluated the benefits of incorporating riparian requirements into environmental flows by estimating the expected repercussions of riparian changes driven by regulated flow regimes on the fish long-term habitat suitability. To this end, the riparian vegetation was modeled for 10-year periods according to three different 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 landscapes. Such ecological modeling approach, where a joint analysis is performed while embracing a suitable time response for the ecosystems involved, enables a realistic biological-response modeling and substantiates the long-term research that is required in environmental flow science (Petts, 2009; Arthington, 2015). 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 (Gippel, 2001; Davies et al., 2013). 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.

During modeling, geomorphology was considered immutable and sediment transport originated by the environmental flow regimes was disregarded. River morphodynamics and its interactions with riparian vegetation constitute an important river process in many rivers, particularly in fine sediment rivers (e.g., Corenblit et al., 2009; Corenblit et al., 2011; Gurnell et al., 2012; Gurnell, 2014). However, the research on the temporal scales of geomorphic and ecological processes is still scarce in coarse-bed rivers (Corenblit et al., 2011), and simultaneously more complex and uncertain (Yasi et al., 2013). The error predictions from best hydraulic predictors in this type of rivers can range between 50 to 200% (Van Rijn, 1993; Yasi et al., 2013). Disregarding such processes in these study sites was carefully considered. Given the above and the arguments mentioned in the methods section, we are confident that this option in this case will not bring tangible shortcomings to this research. Furthermore, the possible riverbed degradation effects due to the releasing of sediment-starving floods by the dam were not tested because according to our expert knowledge this will not pose a

problem in this case. Such floods with similar recurrence intervals were already tested by Rivaes et al. (2015) in two river stretches of much smaller grain size (pebbles and sand) and results showed in both cases that such flood discharges were not relevant for riverbed degradation. The influence of fish species on geomorphology and riparian vegetation by ecosystem engineering, as it was mentioned in the introduction, was not considered also during this study as it seemed fairly unrealistic in these case studies due to the general dimension of riverbed particles.

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. This is an important outcome that is essential to remember when providing environmental flow instructions. Subsequently, microhabitat analysis demonstrated that changes in the riparian landscape induce modifications in the hydraulic characteristics of the river stretches. The differences in mean values of these parameters are subtle between riparian landscapes but are statistically significant. Furthermore, 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 in some places. The hydrodynamic modeling results show that the water flowing near the margins is more affected than the water flowing in deeper areas of the river channel. One reason for these results is certainly because this study is about the effects of riparian vegetation encroachment on the physical habitat due to the colonization of the river margins by woody riparian vegetation.

Accordingly, there are locations where the considered hydraulic parameters change considerably, shifting the habitat preference of fishes in one or two classes of the corresponding habitat preference curves. 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 (Appendix E – Figure E10, Figure E11 and Figure E12).

The hydrodynamic modeling also indicated changes directly affecting the habitat suitability of the existing fish species according to the riparian landscape. Through time,

the riparian landscape shaped by the Eflow regime diverged in habitat suitability from the natural and Eflow&Flush landscapes, 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., Pusey et al., 1993; Matthews, 1998; Clark et al., 2008) and can have a significant impact on the ecological status and function of the existing fish communities (Jones et al., 1996; Randall and Minns, 2000; Freeman et al., 2001). 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 (Gomes-Ferreira et al., 2005; Doadrio, 2011), 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 relationships between fish assemblages and habitat are extremely complex (e.g., Hubert and Rahel, 1989; Diana et al., 2006; Santos et al., 2011), being a consequence of the actual natural conditions (Poff and Allan, 1995; Poff et al., 1997) 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 landscape and consequently, the maintenance of the fish habitat suitability. Accordingly, the implementation of such measure in place of using environmental flows addressing only fish requirements can provide significant positive ecological effects in downstream reaches (Pusey and Arthington, 2003; Lorenz et al., 2013) and additional ecosystem services like stream bank stability, flood risk reduction or wildlife habitat (Blackwell and

Maltby, 2006; Berges, 2009) 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 addition, model uncertainty due to estimation uncertainty in input parameters was previously assessed by means of sensitivity analyses on both models. In either case the models showed to be quite robust to the uncertainty of estimated parameter inputs (see Rivaes et al. 2013, and Boavida et al. 2013b) which reveal a relatively small uncertainty in the models outputs and provides additional confidence on the results.

In conclusion, we predict a change in fish habitat suitability according to the long-term structural adjustments that riparian landscapes endure following river regulation. These changes can be attributed to the effects that altered riparian landscapes 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, 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://doi.org/10.5281/zenodo.839531. 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.

Acknowledgments

SECTION VI

GENERAL DISCUSSION

CHAPTER 8

Discussion and conclusions

1. Discussion of the results

This thesis has the main focus of developing the knowledge on the relationships between river flow regime, riparian woodlands, and remaining aquatic biota and fluvial processes. On this path, and in order to achieve the proposed objectives presented in the beginning of this thesis, it is structured in such a way that it can address several topics that are needed to support its conclusions. Nonetheless, the presented research is backed up by other preparatory works, performed previously or during the development of this thesis, which were not included. For instance, to model the response of riparian vegetation to the river flow regime it was necessary to find a model with sufficient accuracy to provide a reasonable confidence in the results for the analysis of the long-term perspective of the river system. With the purpose of doing so, at the beginning of the thesis, a team of experts met up to ascertain the capacity of the *CASiMiR-vegetation* model (Benjankar et al., 2011) in providing useful and confident results to address these questions. Therefore, the *CASiMiR-vegetation* model, which is the basis for the modeling of riparian vegetation in this thesis, needed to be tested in different circumstances and particularly in Med-rivers. Additionally, to implement the model from scratch it was also necessary to understand and describe the succession-retrogression pathways of riparian vegetation ecological succession in Med-rivers. This potential implementation analysis was published previously by (García-Arias et al., 2013) and constitutes a foundation of this thesis. The interactions between riparian vegetation and fluvial hydraulic processes within the *CASiMiR-vegetation* model were also assessed by Egger et al. (2013) during the beginning of the thesis and backed up the choice for using this model.

This thesis presents the first time ever this model was implemented in the Mediterranean basin, demonstrating its ability to predict the spatiotemporal changes in riparian vegetation guilds in semi-arid river basins. In the course of time, *CASiMiR-vegetation* proved to be a valuable tool to model the response of riparian vegetation to different flow regime scenarios in Med-rivers. Within the scope of the thesis, this tool was applied on several case studies, in regulated and unregulated rivers, achieving good calibration and validation results in all of them. Model accuracies attained a quadratic weighted kappa in the range of 0.61 to 0.69, when perceived as proportion of a maximum possible given the observed marginal frequencies, revealing a good

agreement strength with observed field data (Landis and Koch, 1977; Altman, 1991; Viera and Garrett, 2005). This shows a very good accuracy of the model for Mediterranean rivers and provides confidence on the results presented in the thesis. Furthermore, the model also revealed not to be too much sensitive to possible estimation errors in shear stress parameter thresholds, which provides additional confidence on the results, as this parameter was the only one estimated by model tuning. All the remaining parameterization of the model was measured during field surveys. Finally, model runs showed that *CASiMiR-vegetation* model forecasted similar patterns to other models for equivalent disturbances, such as the behavior of phreatophytes species facing river flow degradation (Lytle and Merritt, 2004), water table drops (Scott et al., 1999), or climate change-driven flow regimes (Dixon and Turner, 2006; de Kok and Booij, 2009; Ström et al., 2012). Consequently, the objective of calibrating of a dynamic vegetation model based on the predictive relationship between riparian flow-response guilds and the river flow regime was achieved.

Despite the inherent stochasticity of fluvial systems, the research presented here adopted a deterministic modeling approach. Although being a simplification, the maximum annual instantaneous river discharges seem to be the ultimate circumstance of morphodynamic forces driving the succession-retrogression dynamics of riparian woodlands (see Junk et al., 1989; Naiman and Décamps, 1997; Tabacchi et al., 1998; Bendix and Hupp, 2000; Friedman and Lee, 2002; Whited et al., 2007b; Stromberg et al., 2010a). This simplification enhances the appreciation of broad features or general trends and allows the understanding on how specific components of the flow regime affect riparian vegetation (see Poff et al. 1997 for a better understanding). Consequently, based on this deterministic approach, we were able to address riparian responses to the discharges that are really important to condition the riparian vegetation dynamics.

Another simplification of the multifaceted complex fluvial processes adopted for modeling was a stable topography. Indeed, river morphodynamics and its interactions with riparian vegetation are considered as an important river process in many rivers, particularly in fine sediment rivers (e.g., Corenblit et al., 2009; Corenblit et al., 2011; Gurnell et al., 2012; Gurnell, 2014). Therefore, disregarding such processes in this thesis was carefully weighted. The research on the temporal scales of geomorphic and ecological processes is still scarce in coarse-bed rivers (Corenblit et al., 2011).

Simultaneously, it is more complex and uncertain (Yasi et al., 2013), presenting error predictions from the best hydraulic predictors in this type of rivers ranging between 50 to 200% (Van Rijn, 1993; Yasi et al., 2013). The interaction between vegetation and sediment transport is even more complex to model (e.g., Gurnell, 2014). In turn, all the case studies considered in this thesis presented fairly stable geomorphologies, occurring in valleys confined to some extent, with bed material alternating generally from bedrock to cobbles. Such conditions may preclude channel movement and sediment transport, resulting in a much more stable river channel with low sinuosity in which broad processes of erosion-sedimentation occur with much more difficulty (Dingman, 2009). Furthermore, the hydraulic analyzes performed in chapter 5 and 7 with this purpose revealed that this was not a noteworthy aspect in these rivers. In this context, it was considered that these complex effects could be disregarded without substantial bias to the modeling results.

The ability to predict riparian patches is a very important aspect for river restoration and management, allowing for the inclusion of the long-term perspective of the fluvial ecosystem (Stromberg et al., 2010b) in the definition of past and future riparian scenarios, environmental flow regimes, environmental directives for reservoir outflow management, river restoration planning, or ecological quality assessment. In this thesis, every modeling of the riparian vegetation considered a period of approximately a decade, which was considered as a suitable time response for the involved ecosystems. Actually, such time scale is sufficient to cover and detect the riparian responses to hydrological and habitat variations (Frissell et al., 1986; Thorp et al., 2008), while avoiding modeling uncertainties due to model constrains. Such ecological modeling approach enables a realistic biological-response modeling and substantiates the long-term research that is required in environmental flow science (Petts, 2009; Arthington, 2015).

1.1.   Section II – Large-scale drivers of riparian vegetation

Understanding how the large-scale drivers of riparian vegetation will influence these ecotones on a near future is a powerful asset to outline management strategies and to appraise the threat potential to which these communities are subject to. By these means, it is possible to obtain the knowledge on how to reverse or attenuate such

threats on regulated rivers. The question if one wants to restore riparian woodlands aiming to the historical natural riverscapes or to the expected climate-changed riverscapes in another matter, which goes beyond the scope of this thesis.

Chapter 2 presents an approach that goes a little further than previous modeling from other authors due to the spatially explicit way in which the results are presented by *CASiMiR-vegetation* model. Such an approach allowed observing that in a natural situation, the spatial distribution of the riparian succession phases in this Med-climate present an age gradient from the inner channel to outwards, with marked characteristics, such as indicator species and probable zonation related to species traits. Additionally, the riparian patches are significantly different, mainly regarding the heights above water table and the age. Other environmental factors may probably be present but with less obvious shaping action on riparian patches (Katz et al., 2005).

The results show that riparian vegetation in Med-rivers will be highly impaired by climate change. Non-woody sparsely vegetated areas are expected to expand outwards due to the area decrease of younger succession phases inside the active channel. Consequently, the recruitment of riparian species will be greatly affected and this will occur most likely due to the increase in flood disturbance during winter combined with an amplified hydric stress during summer and a lost synchrony with hydrological processes (Mahoney and Rood, 1998; Stella et al., 2006) that will determine a lower survival rate and more fragile patches (Hargreaves and Allen, 2003). On the other hand, older succession patches expand inwards from the uplands as a result of the overcoming by more hydric stress tolerant and well adapted species that can establish successfully in the outer parts of floodplain zones. In the more pessimistic scenario of extreme climatic change, it is likely that younger succession phases face total annihilation in Med-rivers. This circumstance may not be so drastic if plants develop ecophysiological adaptations that ameliorate declines in base flows arising from increased aridity (Stromberg et al., 2010a). Nevertheless, if riparian species will be able to perform those strategies in a proper pace coincident with the climate change rate is still an open question. Moreover, other anthropic or natural-driven changes on different scales may also interact with the climate, thus increasing uncertainty about how vegetation will develop in future.

Chapter 3 goes beyond the scope of those earlier results and analyzes Med-rivers in the European context. This was the first time that a joint effort to ascertain the spatiotemporal response of riparian ecosystems to climate-changed flow regimes was made on a European scale. Again, results show that riparian vegetation is highly influenced by the flow regime but changes are more or less notorious according to the nature of the flow regime. The considered case studies encompass the three main water alimentation forms of European rivers (Pardé, 1955; L'vovich, 1979; Wrzesiński, 2013) and therefore can be used as a first approach for the expected changes of riparian woodlands in Europe.

Results show that in an actual natural scenario, the riparian successional age sequence in Med-rivers is similar to the temperate rivers and therefore native trees in the Med-region must have developed strategies for overcoming the constraints of such harsh Mediterranean environment. Nonetheless, the first succession phases tend to be relatively larger than the corresponding succession phases in temperate rivers (Friedman et al., 1996), probably due to the increased shear stress of the Mediterranean flash floods and the harder survival during summer. In a climate change scenario, every flow regime in Europe will be affected. However, accordingly to the flow regime, these changes will be more or less notorious in the response of riparian vegetation, mainly due to the changes in the flood disturbance and hydric stress which are the most important factors of riparian dynamics (Tabacchi et al., 1998; Johnson, 1999; Bendix and Hupp, 2000; Stromberg et al., 2010a).

The Med-region presents a greater uncertainty in pluvial flow regimes associated to the rainfall forecasts during winter (Alcamo et al., 2007b), but all the scenarios are coherent in forecasting lower river flows and accentuated hydric stresses during summer. Yet, between scenarios there will be significant differences in morphodynamic disturbance.

In a climate change scenario, the riparian woodlands will face contradictory changes according to the flow regime. In snow-powered flow regimes, the succession of vegetation will be the prevailing process while in pluvial flow regimes situation will be more complex, mainly with retrogression inside channel and succession in the outer parts of the river. For flow regimes falling between this spectrum results are not linearly correlated to any of the imposed stresses (e.g., Auble and Scott, 1998; Johnson et al.,

2000) and can present a wide range of riparian changes where shear stress and hydric stress do not explain succession dynamics by themselves. The extent of modifications will be different as well, across the considered flow regimes. In nivo-glacial and mountain-fed flow regimes the changes are moderate but in Mediterranean flow regimes, results show not mistake, med-rivers will be the most impaired by climate change and riparian vegetation modifications will most probably be substantial with dramatic changes.

1.2.   *Section III – Small-scale drivers of riparian vegetation*

Despite the large-scale drivers that affect riverine communities at a watershed or river landscape scale, small-scale drivers influence the aquatic and riparian habitats controlling locally the spatial dynamics of the fluvial communities. Chapter 4 is centered on enlightening how specifically these small-scale drivers affect floodplain vegetation for an improved understanding about riparian dynamics and a better management of regulated rivers. More in detail, this is a research on how the landscape features of Mediterranean riparian vegetation, specifically the location and shape of the patches, are driven on a local scale. Herein, the existing theories about the local drivers of riparian vegetation and its succession phases were tested against observed data at four different Med-rivers. Models were successfully fitted but presented different capabilities in explaining the variability of the location and shape of the patches. At a first sight, this reveals that some patch features have different reliance on the considering disturbances. Nevertheless, this study shows with an outstanding explanatory power for the normal pattern of ecological studies (Low-Décarie et al., 2014) that the location of the patches are almost exclusively driven by groundwater hydrology. The morphodynamic disturbance did not have even a significant influence on the location of the patches. This applies to the riparian woodland in general, but also to each succession phase in particular. The only effect that morphodynamic disturbance has in the location of the riparian patches appear to be in the prevention of the vegetation encroachment by particular succession phases. This was an astonishing finding as the morphodynamic disturbance is in general considered to be higher in central channel areas and influencing particularly early succession phases. At least, this is the main current of thought, influenced by a greater research effort in

temperate rivers. However, in small/medium size watersheds, like the majority of Med-rivers, the largest portion of the sediment total load is carried by frequent floods, occurring once or twice a year, being more competent in the lateral erosion of river banks and therefore more stressing to these particular succession phases than in central areas (e.g., Wolman and Miller, 1960).

Regarding the shape of the patches, the considered disturbances were not found significant in driving this feature. The model presented a very low explanatory power and although common in ecological models (Møller and Jennions, 2002), this means that other factors have greater influence in the shaping of the riparian patches. Therefore, this remains an open question as other possible factors have been suggested as drivers of this feature (e.g., Fernandes et al., 2011; Aguiar et al., 2016; Zhou et al., 2016).

Accordingly, it appears that in our typically Mediterranean study sites, the location of the succession phase patches result rather from a zonation driven by water scarcity than from an ecological succession conducted by different fluvial disturbances. In contrast, the landform processes are considered to be the main driving factor of riparian vegetation in temperate and tropical river systems (see Stromberg et al., 1996; Scott et al., 1999; Shafroth et al., 2000; Steiger et al., 2005; Stromberg et al., 2007a). Thus, this circumstance seems to be one more particularity of Mediterranean riparian woodlands, where seasonal water scarcity is common and the riparian ecosystems have similar adaptations and strategies to cope with the particular stresses of analogous Med-rivers (Bonada and Resh, 2013). These results are coherent with previous observations in Med-rivers (e.g., Stromberg et al., 1996; Scott et al., 1999; Shafroth et al., 2000; Stromberg et al., 2007a), but stand as a novelty by quantifying the importance of each fluvial disturbance in the landscape features of riparian vegetation. Moreover, the outcome of this study is useful also for management purposes. This shows that, along with the necessary maintenance of morphodynamic disturbance, the maintenance of a minimum river discharge capable of sustaining water table levels is also necessary to prevent the rearrangement of the riverscape mosaics following flow regime changes or the prevention of the invasion by exotic species (Planty-Tabacchi et al., 1996).

Finally, the research performed in this section, together with the previous section, grant the achievement of the thesis objective of assessing the main drivers of riparian vegetation ecological succession and the evaluation of its relative influence towards the determination of the flow regime requirements of riparian vegetation.

1.3. *Section IV – Riparian vegetation management*

Once a better understand of the riparian vegetation drivers was achieved, it was important to look for ways of managing this ecotone by means of flow regime management, using artificially activated river disturbances. But first, it was needed to find out how to assess the flow requirements of vegetation and how to evaluate the degradation of floodplain areas. Conversely, it was also necessary to understand how these flow requirements could be used for channel maintenance and against nuisance species.

In chapter 5, the development of the vegetation patches was simulated, once again using the *CASiMiR-vegetation* model. According to this modeling, it was possible to confirm the greater variability that the natural flow regime brings to the fluvial patch mosaic when compared to any other regulated flow regime. This consequence can be mainly attributed to the great inter-annual variability of natural Med-rivers, which is very important to riparian vegetation (Pettit et al., 2001; Stella et al., 2013) and is still not being considered in restoration measures downstream of dams. Furthermore, results show that inside the active channel, the patch retrogression, and thus the prevention of channel encroachment, is driven by flood events with a recurrence interval greater than the biannual maximum discharge. The assessed flood frequencies were not the same in the two considered study sites, which may indicate that such assessments are must probably watershed specific. Accordingly, environmental flow regimes addressing riparian requirements should never be determined by general rules of thumb. The implementation of such flood frequencies downstream of dams corresponds to the artificial creation of the different metastable, oscillation and acyclic process types resulting from the impact of disturbances on vegetation succession and retrogression in natural rivers (Formann et al., 2013). Without this disturbance, vegetation encroachment can settle in less than a decade. This prediction is coherent with studies where riparian forests were monitored to assess their response to flow

regulation downstream of a dam (e.g., Shafroth et al., 2002; Braatne et al., 2007; Bejarano et al., 2011; Bejarano and Sordo-Ward, 2011; Benjankar et al., 2012; Egger et al., 2012) and demonstrates the capacity of *CASiMiR-vegetation* to predict riparian responses to flow regime management, as well as the benefit of including riparian communities in environmental flow approaches. Accordingly, such an approach appears to be a good solution for linking riparian vegetation and river restoration (Gumiero et al., 2013), recurring to flood events such as the ones predicted by the *CASiMiR-vegetation* model (Scott et al., 1997; Friedman and Lee, 2002; Braatne et al., 2007; Richter and Thomas, 2007; Peake et al., 2011; Ibáñez et al., 2012) and making environmental flows play an effective role in preventing and reversing encroachment (Miller et al., 2013), which is the main source of ecological integrity lost and impoverishment in riparian ecosystems (Tharme, 2003).

During this investigation, the threat of channel degradation through erosional processes was set aside as the performed sediment transport analyzes to the imposed discharges revealed a harmless geomorphic impact. In fact, the prescribed discharges were only able to entrain accumulated fine sediment, which in fact contribute favorably to the maintenance of river channel and habitat of aquatic communities (e.g., Sullivan and Watzin, 2010). In the long-term, these channel erosion revealed to be derisory.

However, the implementation of such flood regimes may not be applicable in many dams. The limiting factor may not be such the amount of water necessary to implement these kind of floods but the fact that dams simply may not be prepared for such releases. This was what it was observed in the two studied dams. The outlet structures must be designed with this in mind: bottom outlets and spillways together must have enough capacity to release riparian maintenance discharges whenever necessary. At last, this research accomplishes the thesis objectives of developing an initial approach to riparian vegetation restoration measures by flow regime management, the setting of reference conditions for environmental flows, and the prediction of structural and functional changes of river communities affected by long-term flow changes.

Chapter 6 shows another modeling exercise to investigate the possibility of setting a minimum annual flow for the management of vegetation in river channels. It was hypothesized that the maintenance of a minimum discharge during summer would create enough flow disturbance to prevent channel encroachment and even demote

the bloom of exotic species. This hypothesis was sustained by the results, which showed that the local physical instream conditions like flow velocity and water depth are the main drivers of growth and distribution of the considered aquatic macrophytes. The implementation of such flow regime would also be effective in demoting the habitat of the exotic invasive *Myriophyllum aquaticum*, therefore decreasing the risk of invasion of this species and favoring the river ecological quality. The habitat of this exotic species cannot be completely removed, but can be reasonably contained to privilege autochthonous aquatic species and strengthen their competitive performance.

This measure can be recommended with a high level of confidence, given that a good level of accuracy in predicting species distribution, was obtained. Moreover, the tested minimum discharge is less than the mean annual discharge during the vegetative period, showing that this can be a feasible procedure to reduce the risk of invasion and favor a more natural species composition. This outcome stands as an improvement for river restoration science, confirming that it is possible to prevent the degradation of natural aquatic vegetation communities without employing the current widely used mechanical methods for controlling aquatic macrophytes in Europe (Hussner et al., 2017) or the chemical methods used worldwide (Champion and Wells, 2014). By these means, it is also possible to avoid the inherent disadvantages of these measures, such as the harm of non-targeted biota or causing sediment resuspension (e.g. (Getsinger, 1998; Habib and Yousuf, 2014). Furthermore, in line with the results of the previous chapters, this would also contribute to recharge water tables affording a hydric stress relieve to riparian communities in regulated rivers and in more severe future climate conditions.

Finally, the research presented in this chapter is a contribute to achieve the objectives of developing an initial approach to riparian vegetation restoration measures by flow regime management, setting reference conditions for environmental flows, and conceiving a preliminary holistic framework for environmental flows by addressing both riparian and aquatic biota requirements.

1.4. Section V – Ecological feedbacks of riparian vegetation management

It is known that riparian vegetation influences aquatic biota and fluvial processes in a plethora of ways (e.g., Pusey and Arthington, 2003; Baxter et al., 2005; Beltrão et al., 2009; Van Looy et al., 2013). Accordingly, one can expect that changes in the former would produce effects on the latter. Nevertheless, the assessment of these ramifications have been seldom assessed and ecology still lacks to clearly identify the links between restoring flows and the corresponding ecological benefits (Miller et al., 2012), particularly regarding long-term monitoring of environmental flow performance (King et al., 2015, and citations herein). This section is therefore dedicated to the investigation of the potential effects of riparian vegetation management on the aquatic biota and river processes. Particularly, chapter 7 evaluates the benefits of incorporating riparian requirements into environmental flows by estimating the expected repercussions of riparian changes driven by regulated flow regimes on the fish long-term habitat suitability. This provides an insight of the expected long-term effects of environmental flows in river ecosystems, which is an essential topic but poorly considered in environmental flow science (Gippel, 2001; Davies et al., 2013). Assessing these long-term consequences unveil the potential remarkable role of riparian vegetation on the support of environmental flows efficiency, which may transform the actual paradigm in environmental flow science of attributing an almost exclusive emphasis on the aquatic ecosystem. Up to date, we are not aware of such a modeling approach ever being used in the appraisal of the long-term efficiency of environmental flow regimes.

The approach was taken from an ecohydraulic point of view and started by modeling the structural response of riparian vegetation facing different environmental flows for a period of a decade. Then, fish habitat availability was computed for each of the resulting riparian habitat scenarios. Results show that the fish habitat availability is expected to change accordingly to the long-term structural adjustments that riparian habitat endure following river regulation. In detail, changes in riparian vegetation landscape along the river channel can change significantly the hydraulic characteristics of the river channel, namely, water depth and flow velocity. Fish fauna have preferences for particular combinations of these hydraulic parameters (Muñoz-Mas et al., 2012; Boavida et al., 2013b) while river processes only occur if specific conditions regarding these parameters are met (Church, 2002). Consequently, such changes

influence the habitat availability of aquatic fauna and the fluvial processes. In the considered case studies, environmental flow regimes disregarding riparian vegetation led to the degradation of riparian habitats, which in turn transformed significantly the aquatic habitat when compared to the reference natural habitat. Considering the existing relationship between fish assemblages and habitat (e.g., (Pusey et al., 1993; Matthews, 1998; Clark et al., 2008)), such changes can lead to a significant impact on the ecological status and function of these communities (Jones et al., 1996; Randall and Minns, 2000; Freeman et al., 2001), being this impact proportional to the enforced habitat change (Cowley, 2008). On the other hand, environmental flow regimes taking into account riparian vegetation requirements were able to maintain the riparian landscape near natural standards, therefore preserving the naturalness of the aquatic habitat for the targeted fish species. Additionally, the benefits on fish habitat of such environmental flow regimes addressing riparian requirements was also discussed in previous chapter 4, also regarding sediment transport and channel maintenance. Accordingly, this shows that riparian vegetation requirements must be considered on environmental flows in order to assure their effectiveness in the long-term perspective of the fluvial ecosystem. Moreover, the implementation of such measure instead of using environmental flows addressing only fish requirements can provide significant positive ecological effects in downstream reaches (Pusey and Arthington, 2003; Lorenz et al., 2013) and additional ecosystem services like banks stability, flood risk reduction or wildlife habitat (Blackwell and Maltby, 2006; Berges, 2009), while imposing minor revenue losses to dam managers as it was observed in a previous chapter. Finally, these results provide the necessary evidence to support the attainment of the thesis objectives regarding the conception and testing of a preliminary holistic framework for environmental flows, the prediction of structural and functional changes of river communities affected by long-term flow changes, and the assessment of the ecological impact of riparian vegetation management on aquatic communities.

1.5. Future research opportunities

Scientific investigation is a never-ending activity. As scientific knowledge expands on a certain subject, new questions appear and open up new research opportunities. This thesis is no exception and, throughout its development new research topics have

emerged, paving the way for improved knowledge about developing topics that can be considered yet insufficiently studied in fluvial ecosystems (Winemiller et al., 2010).

Accordingly, despite the recent recognition of the issues concerning the modeling of interactions between flow regime, vegetation and morphology (Gurnell et al., 2012; Camporeale et al., 2013), such processes were not yet implemented in the *CASiMiR-vegetation* model and call for a specific research effort aiming at their integration in future model developments (Camporeale et al., 2013). In fact, the development of suitable models to simulate and analyze the biogeomorphologic feedbacks is still a priority in ecogeomorphology science agenda (Corenblit et al., 2011), as limited capacity remains to predict flow properties in vegetated channels, due to the great difficulty of linking complex dynamic vegetation structures to non-homogeneous hydrogeomorphic processes (Corenblit et al., 2007).

Chapter 4 was inconclusive about the small-scale drivers affecting the shape of the patches. Hence, a similar approach encompassing a broader range of possible drivers could enable the discovery of the main factors steering this feature.

The results obtained in this thesis by modeling vegetation dynamics generated further new questions stemming from riparian ecology concepts. The expected changes in the spatial ratio of different riparian types, with the likely suspension of succession in some cases, could lead to reflect on the interplay between the fluvial setting and vegetation (e.g., Muneepeerakul et al., 2007) – i.e. the relative dominance of non-equilibrium *versus* quasi-equilibrium processes (Bendix and Hupp, 2000). This thesis also suggests new scientific questions regarding the potential feedbacks of novel habitats associated with an altered riparian vegetation mosaic, leading to changes in shear stress disturbance and hydrogeomorphic processes (Gran and Paola, 2001; Johnson, 2002), or in relation to potential alterations in the global functioning of the ecosystem and thus the services it provides.

Finally, it was an objective of this thesis to conceive and evaluate a preliminary holistic framework for environmental flows. This objective was considered to be accomplished in the sense that it was able to jointly address together intra- and interanual requirements of the fluvial ecosystem. However, the term holistic can only be properly used if, in fact, the whole river ecosystem is addressed. For this, several other issues must be considered in further investigation. For instance, the presented framework still

just incorporates one biologic element of aquatic fauna. Although one can argue that fish fauna can stand for the aquatic requirements in general, the fact is that macroinvertebrate requirements should be assessed too, even if only to confirm the appropriateness of the fish results into macroinvertebrate habitat. Furthermore, although sediment transport modeling was considered in the present work, more detailed investigation could be performed to fully address this issue in this environmental framework. Several other topics must be analyzed and tested in this approach, such as thermal or chemical regimes, groundwater dependencies, estuarine responses or even cultural demands (e.g., Livingston et al., 1997; Boulton, 2007; Nilsson and Renöfält, 2008; Olden and Naiman, 2010; Magdaleno, 2018). In the end, an active long-term monitoring and assessment of the established environmental flows along with an adaptive revision based on monitoring is likewise necessary (Stanford et al., 1996) for an effective river restoration.

2. Conclusions

Based on the work performed, embodied by the published papers that compose each chapter of this manuscript, one can conclude that:

- A first performance test of the riparian vegetation dynamic model *CASiMiR-vegetation* in the Mediterranean basin was performed, demonstrating its ability to predict the spatiotemporal changes of riparian vegetation guilds in this type of climate;
- The calibration and validation of the *CASiMiR-vegetation* model was successfully achieved with substantial strengths of agreement and robustness in dealing with uncertainty due to parameter estimation errors.

Regarding the large-scale drivers of riparian vegetation:

- In Mediterranean areas, climate change will affect river flow regimes and cause riparian vegetation amendments along rivers, with a particular general reduction of the riparian vegetation areas;

- Everywhere, younger and more water-dependent succession phases of riparian vegetation will be the most affected by climate change;
- There can be dramatic modifications in the riparian landscapes, posing serious threats for the viability of particular species populations with important conservation issues;
- The Mediterranean rivers are expected to be the most imperiled in Europe by climate change.

Regarding the small-scale drivers of riparian vegetation:

- The fluvial disturbances were demonstrated to have different effects on the location and shape of riparian vegetation patches;
- The main driver of riparian patch location was groundwater hydrology and indicated a predominant zonation of riparian succession phases over natural ecological succession;
- The morphodynamic disturbances are nevertheless still responsible for preventing vegetation encroachment;
- The patch shape seemed not to be primarily driven by the considered fluvial disturbances but, within the limited explained variability of the proposed model, both groundwater hydrology and morphodynamic disturbances have a substantial impact;
- There is a likely necessity for specific procedures during flow regime management to account for the particularities of riparian vegetation in Mediterranean rivers.

Regarding the riparian vegetation management:

- It was possible to successfully model the expected natural riverscape in regulated rivers, hence enabling setting reference conditions for environmental flows in regulated rivers;
- Adequate flushing flows have the ability to restore riparian patch dynamics and reduce regulation effects on riparian communities downstream of dams, enabling the development of an initial approach to riparian vegetation restoration measures by flow regime management;

- The proposed flushing flows did not cause significant erosion on fluvial geomorphology, for the test cases used;

- The various effects of similar flow regimes on both case studies indicate that the establishment of guidelines for environmental flow regimes considering riparian requirements should only be applicable at most to the watershed scale;

- A careful planning of environmental flows using a holistic perspective, which should also include riparian requirements, must be based on a numerical modeling approach, preferably preceding dam design, so that the dam outlet structures can meet flow requirements;

- The growth and distribution of macrophytes is a function of the local instream conditions;

- Results provide a great confidence in the possibility of reducing the risk of invasion by exotic species through the management of flow regimes;

- The maintenance of a minimum flow regime to prevent vegetation encroachment in regulated rivers implies during the vegetative period a discharge smaller than the mean flow of the corresponding period;

- Setting minimum discharges may not prevent completely an invasion by exotic species, but can surely privilege autochthonous species and strengthening their competitive performance.

Regarding the ecological feedbacks of riparian vegetation management:

- The natural flow regime generates morphodynamic disturbances, without which the riparian vegetation is able to settle and age in the river channel;

- It was possible to predict structural and functional changes of riparian vegetation affected by long-term changes in the natural flow regime;

- Riparian vegetation withstands structural long-term changes when facing river regulation or environmental flows disregarding its flow requirements;

- Fish habitat suitability changes according to the long-term structural adjustments that riparian landscapes endure following river regulation;

- Fish habitat changes could be attributed to the effects that altered riparian landscapes have on the hydraulic characteristics of the river stretches;

- 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;

- Without considering riparian requirements, environmental flows are unsustainable in the long term, 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;

- Accounting for riparian vegetation requirements is an essential measure to assure the effectiveness of environmental flow regimes in the long-term perspective of the fluvial ecosystem.

Final comments towards the analysis of the success in attaining the thesis objectives:

Based on the discussion and conclusions presented in this chapter, it is argued that all the objectives of this thesis were accomplished in a satisfactory manner, namely:

- The objective of calibrating and validating a dynamic vegetation model based on the predictive relationship between riparian flow-response guilds and the river flow regime was achieved with the research presented in chapters 2, 3, 5 and 7.

- The objective of developing an initial approach to riparian vegetation restoration measures by flow regime management was accomplished by chapters 4, 5, 6 and 7;

- The objective of setting reference conditions for environmental flows in regulated rivers was reached by chapters 4, 5 and 6;

- The objective of conceiving a preliminary holistic framework for environmental flows by combining both riparian and aquatic elements was achieved in chapter 7;

- The objective of testing and validating the preliminary holistic framework in river reaches presenting different types of flow regulation was reached by chapter 7;

- The objective of predicting the structural and functional changes of river communities affected by long-term flow changes was attained in chapter 7;

- The objective of evaluating the ecological feedbacks of riparian vegetation management on aquatic communities was accomplished in chapter 5 and 7.

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APPENDICES

APPENDIX A

Table A1. Confidence intervals for mean shear stress differences between scenarios in each case study.

Case study	Reference – Optimist		Reference – Pessimist		Optimist – Pessimist	
	Confidence interval	p-value	Confidence interval	p-value	Confidence interval	p-value
Kleblach reach	[6.38;6.61]	< 2.2e-16	[1.69;1.92]	< 2.2e-16	[-4.81;-4.57]	< 2.2e-16
Ribeira reach	[-4.06;-2.11]	3.248e-16	[12.98;13.52]	< 2.2e-16	[15.36;17.32]	< 2.2e-16
Terde reach	[2.04;2.37]	< 2.2e-16	[2.37;2.69]	< 2.2e-16	[0.17;0.48]	6.435e-8

Confidence intervals have a confidence level of 99%.
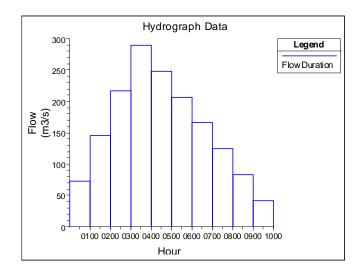
APPENDIX B

Figure B1. Upstream cross-section flow series considered as boundary conditions type in quasy-unsteady flow data for the Odelouca case study.
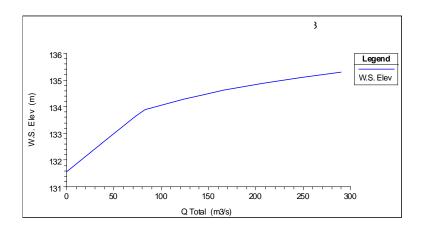

Figure B2. Downstream boundary condition for the Odelouca case study.
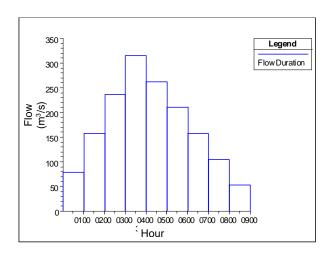
Figure B3. Upstream cross-section flow series considered as boundary conditions type in quasy-unsteady flow data for the Monte da Rocha case study.
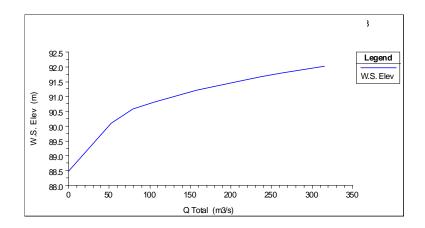
Figure B4. Downstream cross-section rating curve considered as boundary conditions type in quasy-unsteady flow data for the Monte da Rocha case study.

Table B1. Considered transport parameters and boundary conditions of sediment data.

		Monte da Rocha	Odelouca
Transport parameters	**Transport Function**	Laursen (Copeland)	Laursen (Copeland)
	Sorting Method	Exner 5	Exner 5
	Fall Velocity Method	Ruby	Ruby
Sediment boundary condition type	**Upstream cross-section**	Sediment time series (null load)	Sediment time series (null load)
	Downstream cross-section	Equilibrium load	Equilibrium load

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Figure B5. Bed gradation of sediment data at Odelouca channel (left) and floodplain (right).

Figure B6. Bed gradation of sediment data at Monte da Rocha channel (left) and floodplain (right).

APPENDIX C

Table C1. Model-calibrated parameters.

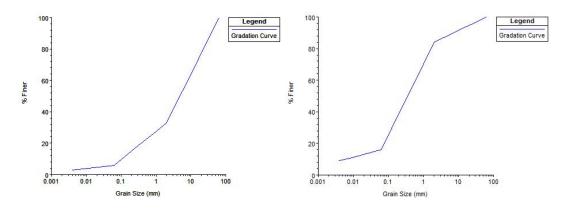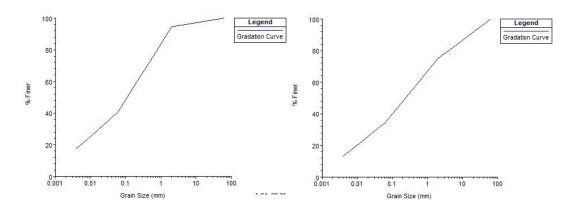| Parameter	Calibration
Succession phase age (years)	0-2
	2-5
	5-16
	16-49
	>49
HBWL for woodland recruitment in the BZ (m)	0.095-1.720
HBWL for woodland recruitment in the FPZ (m)	0.095-1.720
HBWL for scour disturbance zone	<0.095
Critical shear stress of woodland (N/m²)	30 (IP)
	30 (PP)
	50 (ES)
	300 (EF)
	300 (MF)

HBWL – Height above base water level; BZ – Bank zone; FPZ – Floodplain zone; IP – Initial phase; PP – Pioneer phase; ES – Early successional woodland phase; EF – Established forest phase; MF – Mature forest phase.

Table C2. Confusion matrix of the comparison between the observed and simulated vegetation maps.

		Modeled				
		IP	PP	ES	EF	MF
Observed	IP	147	0	93	0	15
	PP	13	0	38	0	0
	ES	1705	6429	35551	8696	946
	EF	4	5565	1621	23869	9542
	MF	66	186	369	15444	14595

APPENDIX D

Table D1. Shapiro-Wilk normality test, by succession phase, with a confidence level of 95% (null hypothesis: population is normally distributed).

Succession phase	Monte da Rocha		Odelouca	
	W	p-value	W	p-value
IP	0.8271	0.0001112	0.93	0.1227
PP	0.8341	0.0001553	0.9088	0.04473
ES	0.7708	9.398e-06	0.8296	0.001511
EF	-	-	0.677	1.012e-05

Table D2. Wilcoxon signed rank test with continuity correction for the Monte da Rocha data with a confidence level of 95% (null hypothesis: true location shift is equal to zero).

Succession phase	Actual scenario *vs.* Natural Scenario		Actual scenario *vs.* Eflow scenario		Eflow scenario *vs.* Natural scenario	
	V	p-value	V	p-value	V	p-value
IP	0	0.003822	0	0.003353	31	0.8938
PP	52	0.01431	35	0.01768	21	0.5403
ES	66	0.003273	66	0.002934	36	0.8191

Table D3. Wilcoxon signed rank test for the Odelouca data with a confidence level of 95% (null hypothesis: true location shift is equal to zero).

Succession phase	Natural scenario *vs.* Eflow scenario	
	V	p-value
IP	46	0.2783
PP	39	0.6247
ES	0	0.003667
EF	0	0.002085

APPENDIX E

Table E1. Flow curve considered in the downstream section of OCBA study site as the outflow condition in River2D model.

Discharge (m³ s⁻¹)	Water surface elevation (m)
0.05	139.65
0.1	139.67
0.3	139.80
0.5	139.92
0.7	140.00
1	140.08
2	140.25
3	140.39
4	140.50
5	140.60

Table E2. Flow curve considered in the downstream section of OCPR study site as the outflow condition in River2D model.

Discharge (m³ s⁻¹)	Water surface elevation (m)
0.1	113.68
0.5	113.77
0.7	113.82
1	113.89
1.5	113.95
2	114
2.5	114.04
3	114.08
4	114.14
5	114.2
6	114.25
7	114.29
8	114.34
9	114.38
10	114.42
12	114.49
14	114.58
16	114.64
18	114.7
20	114.76
25	114.9
30	115.03
40	115.25
50	115.42
60	115.55
80	115.78
100	116
150	116.47
200	116.82
300	117.41
400	117.96
600	118.71
800	119.28
1000	119.77
1200	120.22

Table E3. Channel roughness classification of the different substrates in the aquatic zone of the river without vegetation used in the River2D model for both case studies.

Substrate	Effective roughness height, k_s (m)
Sand and gravel	0.1
Boulders and stones	0.5

Table E4. Channel roughness classification of the different considered riparian vegetation succession phases used in the River2D model for both case studies (IP – Initial phase, PP – Pioneer phase, ES – Early succession woodland phase, EF – Established forest phase and MF – Mature forest phase).

Succession phase	Effective roughness height, k_s (m)
IP	0.4
PP	0.5
ES	1.6
EF	1.1
MF	1.4

Table E5. *CASiMiR-vegetation* model parameterization (IP – Initial phase, PP – Pioneer phase, ES – Early succession woodland phase, EF – Established forest phase and MF – Mature forest phase).

Parameter	Succession phase	Value
Height to water table elevation (m)	IP	< 0.2
	PP	0.2 – 0.6
	ES	0.6 – 1.05
	EF	1.05 – 3.4
	MF	> 3.4
Age (years)	IP	< 2
	PP	2 – 6
	ES	6 – 19
	EF	19 – 26
	MF	> 26
Resistance to shear stress (N m^{-2})	IP	30
	PP	30
	ES	50
	EF	300
	MF	300

Table E6. Patch characterization of succession phases (IP – Initial phase, PP – Pioneer phase, ES – Early succession woodland phase, EF – Established forest phase and MF – Mature forest phase).

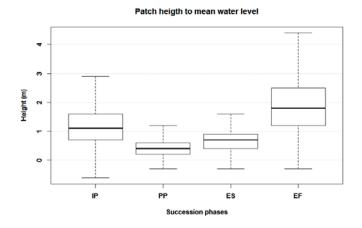| Succession phase	Patches surveyed	Mean height to mean water level (m)	Mean area (m^2)	Mean cover of herb layer (%)	Mean cover of shrub layer (%)	Mean cover of tree layer (%)	Mean number of woody species
IP	11	1.12	357.51	0.48	0.05	0.00	0
PP	17	0.40	350.73	0.81	0.26	0.00	1
ES	20	0.68	256.82	0.29	0.71	0.04	2
EF	8	1.89	1132.20	0.61	0.46	0.19	5

Figure E1. Patch height to mean water level grouped by succession phase (IP – Initial phase, PP – Pioneer phase, ES – Early succession woodland phase and EF – Established forest phase).

Figure E2. Patch age grouped by succession phase (IP – Initial phase, PP – Pioneer phase, ES – Early succession woodland phase and EF – Established forest phase).

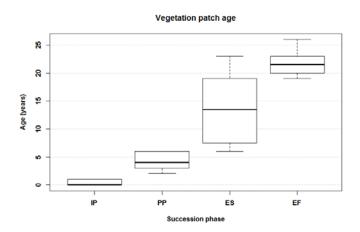Table E7. Number of captured cyprinid individuals throughout different sampling seasons in Ocreza river basin.

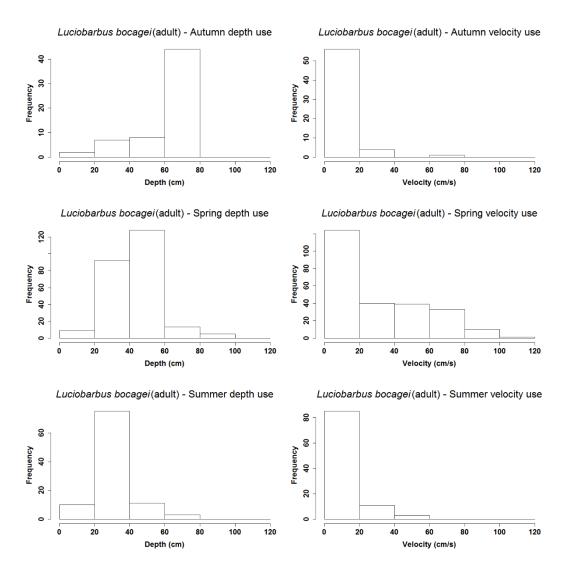| Common name	Cyprinid Species	Spring	Summer	Autumn	Total
Iberian barbel	*Luciobarbus bocagei*	666	300	102	1068
Iberian straight mouth-nase	*Pseudochondrostoma polylepis*	46	62	102	210
Calandino	*Squalius alburnoides*	277	364	134	775
Southern Iberian chub	*Squalius pyrenaicus*	9	0	29	38
Total		998	726	367	2091

Figure E3. Use frequency of Luciobarbus bocagei adults for water depth (cm) and flow velocity (cm s^{-1}) during Autumn, Spring and Summer.
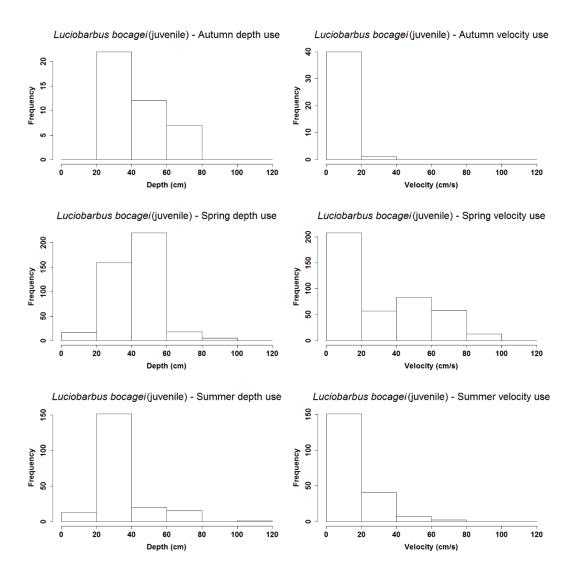
Figure E4. Use frequency of *Luciobarbus bocagei* juveniles for water depth (cm) and flow velocity (cm s^{-1}) during Autumn, Spring and Summer.

Figure E5. Use frequency of *Pseudochondrostoma polylepis* adults for water depth (cm) and flow velocity (cm s^{-1}) during Autumn, Spring and Summer.

Figure E6. Use frequency of *Pseudochondrostoma polylepis* juveniles for water depth (cm) and flow velocity (cm s^{-1}) during Autumn, Spring and Summer.

Figure E7. Use frequency of *Squalius alburnoides* adults for water depth (cm) and flow velocity (cm s⁻¹) during Autumn, Spring and Summer.
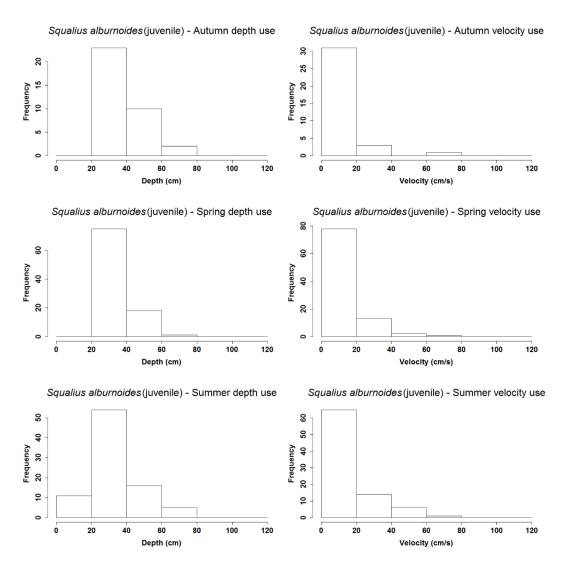
Figure E8. Use frequency of *Squalius alburnoides* juveniles for water depth (cm) and flow velocity (cm s⁻¹) during Autumn, Spring and Summer.
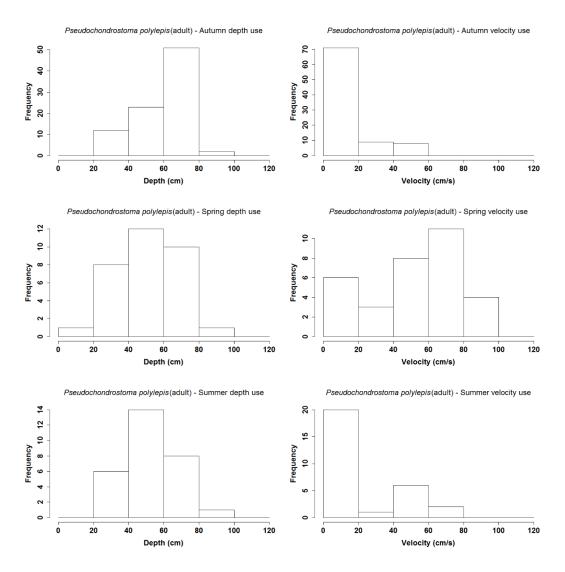
Figure E9. Habitat availability for water depth (cm) and flow velocity (cm s^{-1}) during Autumn, Spring and Summer.

Figure E10. Habitat preference of *Luciobarbus bocagei* for water depth (cm) and flow velocity (cm s^{-1}) during Autumn, Spring and Summer.

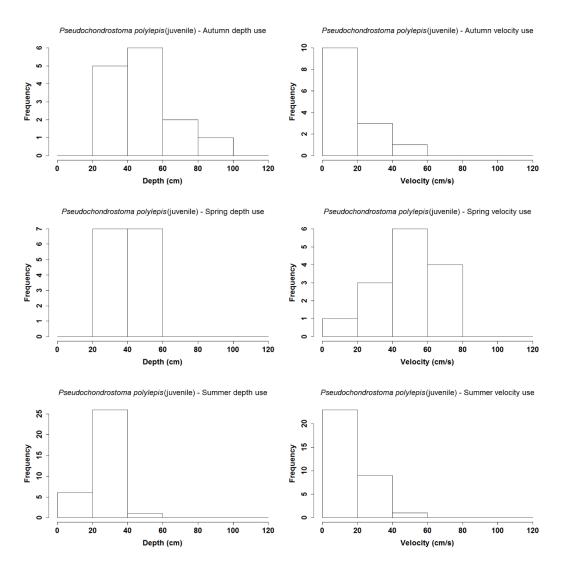Figure E11. Habitat preference of *Pseudochondrostoma polylepis* for water depth (cm) and flow velocity (cm s^{-1}) during Autumn, Spring and Summer.

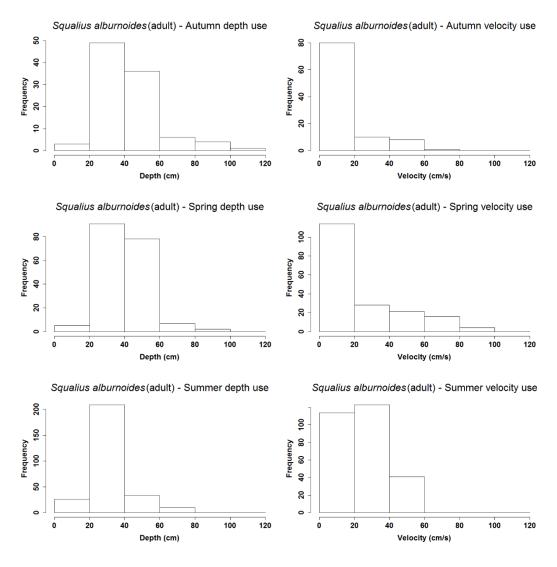Figure E12. Habitat preference of *Squalius alburnoides* for water depth (cm) and flow velocity (cm s^{-1}) during Autumn, Spring and Summer.
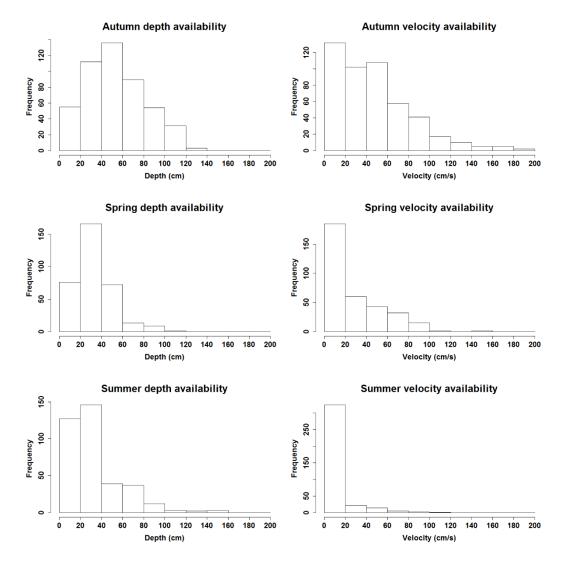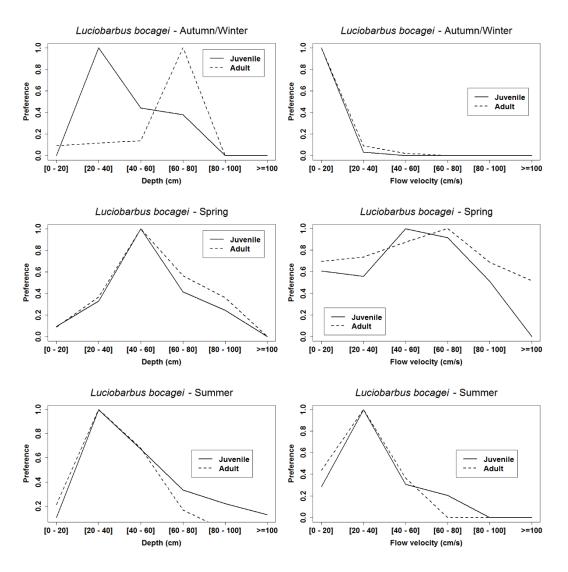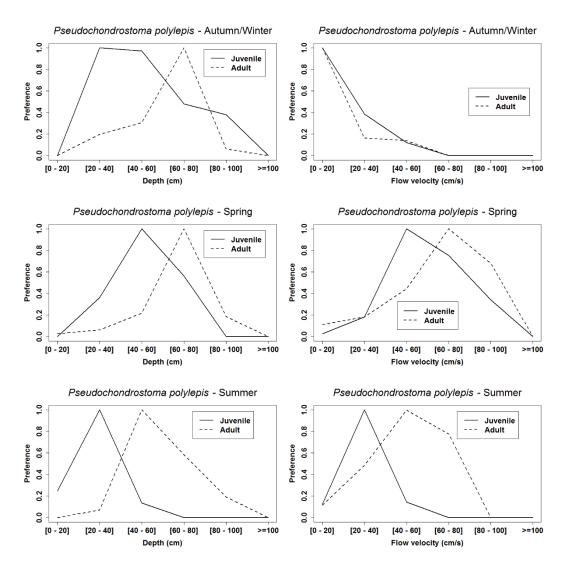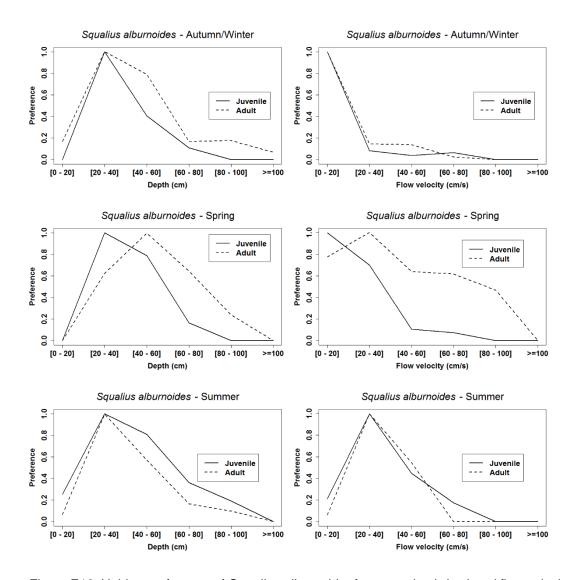
Table E8. Results of the t-tests (H0: true difference in means is equal to 0) for the differences between Natural, Eflow and Eflow&Flush habitats in channel roughness for OCBA and OCPR study sites.

	OCBA			OCPR		
	t	df	p-value	t	df	p-value
Natural-Eflow&Flush	-94.978	124890	< 2.2E-16	-5.511	28189	3.6E-08
Natural-Eflow	-194.420	118850	< 2.2E-16	-66.604	27816	< 2.2E-16
Eflow&Flush-Eflow	-92.292	137650	< 2.2E-16	-61.231	27855	< 2.2E-16

Table E9. Results of the t-tests (H0: true difference in means is equal to 0) for the differences between Natural, Eflow and Eflow&Flush habitats in water depth for OCBA and OCPR study sites.

	OCBA			OCPR		
	t	df	p-value	t	df	p-value
Natural-Eflow&Flush	-2.047	445600	0.0407	0.107	121100	0.9145
Natural-Eflow	-5.841	448080	5.2E-09	-1.545	121360	0.1222
Eflow&Flush-Eflow	-3.789	450190	1.5E-04	-1.653	121500	0.0983

Table E10. Results of the t-tests (H0: true difference in means is equal to 0) for the differences between Natural, Eflow and Eflow&Flush habitats in flow velocity for OCBA and OCPR study sites.

	OCBA			OCPR		
	t	df	p-value	t	df	p-value
Natural-Eflow&Flush	2.652	445460	0.0080	0.354	121090	0.7230
Natural-Eflow	16.122	443090	< 2.2E-16	3.629	121060	2.8E-04
Eflow&Flush-Eflow	13.457	446670	< 2.2E-16	3.279	121290	1.0E-03

Table E11. Equality of proportions between Natural habitat and Eflow habitat in OCBA study site (H0: WUA's have the same true proportion). Species codes stand for Lb – *Luciobarbus bocagei*; Pp – *Pseudochondrostoma polylepis*; Sa – *Squalius alburnoides*.

Month	Lb_juv	Lb_adult	Pp_juv	Pp_adult	Sa_juv	Sa_adult
Oct	0.000216	0.582952	6.21E-05	0.940216	0.323139	0.297538
Nov	0.000148	0.530769	1.74E-05	0.896214	0.370834	0.329465
Dec	0.000608	0.244612	1.17E-07	0.483554	0.668862	0.868717
Jan	0.000519	0.257775	1.41E-07	0.510652	0.789124	0.991580
Feb	0.000229	0.388867	1.60E-06	0.723955	0.587666	0.438736
Mar	0.005353	0.053108	0.462443	0.688765	0.035578	7.11E-05
Apr	0.005855	0.037780	0.398626	0.717639	0.041171	7.97E-05
May	0.001723	0.709184	0.524064	0.379709	0.005987	1.65E-05
Jun	0.795967	0.142917	0.610609	0.548629	0.248239	0.823737
Jul	0.878494	0.902296	0.931916	0.887568	0.978652	0.745271
Aug	0.878494	0.902296	0.931916	0.887568	0.978652	0.745271
Sep	0.308822	0.576689	0.272860	0.683524	0.562069	0.474081

Table E12. Equality of proportions between Natural habitat and Eflow&flush habitat in OCBA study site (H0: WUA's have the same true proportion). Species codes stand for Lb – *Luciobarbus bocagei*; Pp – *Pseudochondrostoma polylepis*; Sa – *Squalius alburnoides*.

Month	Lb_juv	Lb_adult	Pp_juv	Pp_adult	Sa_juv	Sa_adult
Oct	0.850180	0.975380	0.937319	0.982124	0.889660	0.929312
Nov	0.837909	0.975984	0.918213	0.986510	0.879203	0.920722
Dec	0.088235	0.361823	0.101082	0.422564	0.247659	0.189515
Jan	0.146970	0.462011	0.162449	0.516233	0.312924	0.256096
Feb	0.721348	0.970973	0.770267	0.981017	0.757865	0.774812
Mar	0.934309	0.853870	0.863878	0.997663	0.934126	0.976693
Apr	0.929003	0.844001	0.858171	0.998189	0.928228	0.970043
May	0.984099	0.955728	0.944510	0.987440	0.993220	0.956985
Jun	0.951283	0.919954	0.985273	0.981322	0.998841	0.911286
Jul	0.891434	0.870963	0.924438	0.982402	0.891084	0.850580
Aug	0.891434	0.870963	0.924438	0.982402	0.891084	0.850580
Sep	0.747660	0.865492	0.690731	0.875758	0.796120	0.742462

Table E13. Equality of proportions between Natural habitat and Eflow habitat in OCPR study site (H0: WUA's have the same true proportion). Species codes stand for Lb – *Luciobarbus bocagei*; Pp – *Pseudochondrostoma polylepis*; Sa – *Squalius alburnoides*.

Month	Lb_juv	Lb_adult	Pp_juv	Pp_adult	Sa_juv	Sa_adult
Oct	0.467174	0.787298	0.047326	0.505018	0.115923	0.066840
Nov	0.018055	0.443040	5.32E-07	0.294739	0.002503	0.001363
Dec	0.002366	0.274062	1.61E-12	0.154162	0.001140	0.000319
Jan	0.002625	0.290223	3.80E-12	0.156759	0.001164	0.000346
Feb	0.008752	0.386039	7.98E-09	0.233811	0.001850	0.000819
Mar	0.061766	0.554872	1.42E-05	0.295786	0.004822	0.002455
Apr	0.001355	0.114244	0.226976	0.716657	0.141175	3.90E-12
May	0.019642	0.262956	0.380338	0.839495	0.170810	1.42E-08
Jun	0.442916	0.665689	0.509405	0.856592	0.751174	0.715195
Jul	0.976778	0.991759	0.958640	0.981950	0.917565	0.869678
Aug	0.918253	0.850503	0.903803	0.915809	0.852783	0.913014
Sep	0.967843	0.907693	0.935341	0.945988	0.884826	0.902486

Table E14. Equality of proportions between Natural habitat and Eflow&flush habitat in OCPR study site (H0: WUA's have the same true proportion). Species codes stand for Lb – *Luciobarbus bocagei*; Pp – *Pseudochondrostoma polylepis*; Sa – *Squalius alburnoides*.

Month	Lb_juv	Lb_adult	Pp_juv	Pp_adult	Sa_juv	Sa_adult
Oct	0.948755	0.972671	0.966995	0.994709	0.997594	0.995991
Nov	0.918275	0.834241	0.982070	0.975136	0.978806	0.989342
Dec	0.948791	0.944823	0.979856	0.998893	0.981760	0.987061
Jan	0.951852	0.953759	0.980720	0.987732	0.994761	0.998211
Feb	0.935172	0.985107	0.982133	0.971475	0.981222	0.986931
Mar	0.953429	0.997549	0.984473	0.987587	0.999956	0.995221
Apr	0.986832	0.997796	0.999768	0.998475	0.988456	0.960142
May	0.992298	0.970561	0.999811	0.998420	0.993079	0.974589
Jun	0.994943	0.990000	0.996416	0.994406	0.994190	0.991033
Jul	0.835681	0.860729	0.846797	0.931103	0.841102	0.835942
Aug	0.808999	0.774405	0.806757	0.882440	0.777273	0.846911
Sep	0.978260	0.817781	0.841934	0.910452	0.819581	0.863367

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