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**A probabilistic approach to quantifying  
spatial patterns of flow regimes and  
network-scale connectivity**

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Silvia Garbin  
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# Sommario

Questa tesi propone un approccio probabilistico per quantificare la distribuzione spaziale della connettività idrologica alla scala di rete dettata dalla variabilità spazio-temporale dei deflussi. Le dinamiche spaziali dei deflussi sono definite sulla base delle caratteristiche climatiche e morfologiche del bacino contribuente, integrando un approccio fisicamente basato che descrive la stocasticità delle precipitazioni, ed incorporando un modello di bilancio idrologico e un modello di deflusso di recessione geomorfologica. La connettività idrologica lungo il reticolo idrografico è valutata sulla base di soglie associate a tiranti idrici ecologicamente significativi. Il modello proposto consente una descrizione quantitativa delle principali cause idrologiche e delle conseguenze ecologiche delle dinamiche dei tiranti sperimentate dalle reti fluviali. Le analisi condotte in questa tesi mostrano che la variabilità spaziale della connettività idrologica alla scala locale dipende fortemente dalla distribuzione spaziale e temporale delle variabili climatiche. I risultati evidenziano come la forma del reticolo connesso possa essere influenzata in modo cruciale dalle dinamiche idrologiche sottostanti. L'approccio a scala di rete sviluppato in questo lavoro fornisce importanti elementi per la comprensione dell'effetto del clima sulla funzione ecologica dei corsi d'acqua naturali e offre una nuova prospettiva per la definizione dei deflussi ecologici determinanti per lo sviluppo di politiche idriche sostenibili.



# Abstract

This thesis proposes a probabilistic approach for the quantitative assessment of reach- and network-scale hydrological connectivity as dictated by river flow space-time variability. Spatial dynamics of daily streamflows are estimated based on climatic and morphological features of the contributing catchment, integrating a physically based approach that accounts for the stochasticity of rainfall with a water balance framework and a geomorphic recession flow analysis. Ecologically meaningful minimum stage thresholds are used to evaluate the connectivity of individual stream reaches, and other relevant network-scale connectivity metrics. The framework allows a quantitative description of the main hydrological causes and the ecological consequences of water depth dynamics experienced by river networks. The analysis conducted in this thesis shows that the spatial variability of local-scale hydrological connectivity strongly depends on the spatial and temporal distribution of climatic variables. Depending on the underlying climatic settings and the critical stage threshold, loss of connectivity can be observed in the headwaters or along the main channel, thereby originating a fragmented river network. The network-scale approach developed in this work provides important clues for understanding the effect of climate on the ecological function of river corridors, and offers a new perspective to define ecological flows for the development of sustainable water policies.





# Contents

<b>1</b>	<b>Introduction</b>	<b>1</b>
<b>2</b>	<b>Methods</b>	<b>7</b>
2.1	Streamflow model . . . . .	7
2.1.1	Rainfall model . . . . .	7
2.1.2	Water balance model . . . . .	10
2.1.3	Recession flow model . . . . .	11
2.1.4	Probabilistic description of streamflows . . . . .	12
2.2	Stage dynamics and connectivity metrics . . . . .	15
2.3	Simulation set-up . . . . .	19
<b>3</b>	<b>Hydrological Connectivity</b>	<b>25</b>
3.1	Effects of climate on network connectivity . . . . .	25
3.2	Spatial variability of hydrological variables . . . . .	29
<b>4</b>	<b>Habitat suitability</b>	<b>37</b>
4.1	Habitat suitability distribution . . . . .	37
4.2	Ecological value of hydrological connectivity . . . . .	39
<b>5</b>	<b>Ecological flows</b>	<b>43</b>
5.1	EU Water Framework Directive . . . . .	43
5.2	From environmental flows to ecological flows . . . . .	51
5.3	A new perspective on the Eflows evaluation . . . . .	55
<b>6</b>	<b>Discussion</b>	<b>59</b>

<b>7 Conclusions</b>	<b>65</b>
<b>Reference</b>	<b>67</b>

# Chapter 1

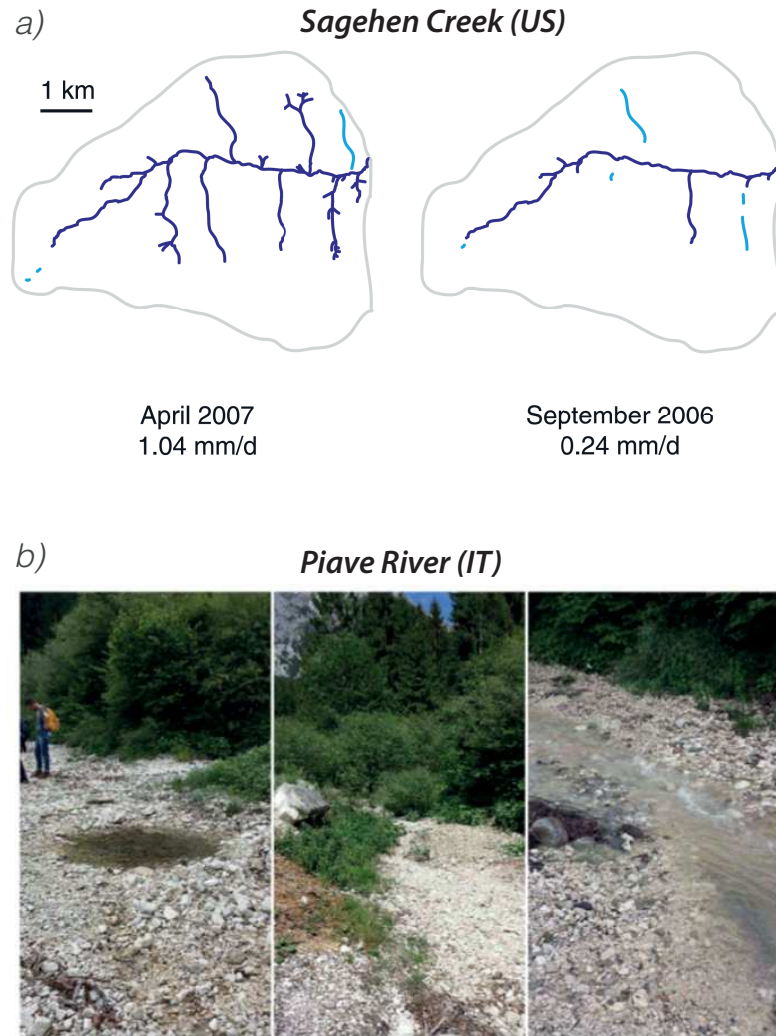
## Introduction

### Motivation

River networks are key elements of the landscape, as they represent ecological corridors for biological species and contribute significantly to shape the hydrological response of catchments [Rodriguez-Iturbe et al., 2009; Biswal and Marani, 2010; Rinaldo et al., 2018]. In the large majority of existing theoretical and experimental works concerning propagation of waterborne diseases, ecological dispersion, and catchment-scale biogeochemistry [Battin et al., 2003; Muneeppeerakul et al., 2008; Gatto et al., 2012; Bertuzzo et al., 2017; Rinaldo et al., 2017], river networks are thought of as static connections between fixed nodes defined on the basis of the topography of the terrain [Mark, 1988; Tarboton et al., 1991; Dietrich et al., 1993; Montgomery and Foufoula-Georgiou, 1993]. However, empirical observations suggest a dynamic behaviour of the flowing network, which is a reflection of the underlying space-and-time variability of hydrological processes. The continuous expansion and contraction of stream width and depth in response to time-variant hydroclimatic forcing (e.g. rainfall) may create physical disconnections between river segments [Godsey and Kirchner, 2014; Jaeger et al., 2014], as shown in Figure 1.1. Therefore, the ecological function of river networks can be significantly reduced by unfavourable local hydraulic conditions that challenge the migration of fishes, propagules and invertebrates, with notable implications for metacommuni-

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ties composition [Tetzlaff et al., 2008; Campbell et al., 2015; Lazzaro et al., 2017; Sarremejane et al., 2017].



**Figure 1.1:** Dynamical nature of river networks. a) Results of the yearly mapping of the drainage network in the Sagehen Creek, Colorado river [Godsey and Kirchner, 2014]. The comparison of the two maps shows the extent of river networks depending on the hydrologic state of the system; b) Change in the hydrologic state of a river reach in the Piave river (Borsoia creek) along the Summer of 2015. The same location is featured by isolated pools (left), dry-bed (center) and flowing water (right).

Climate, vegetation and landscape properties determine the natural flow regime of rivers upon which its ecological integrity depends [Poff et al., 1997; Allan and Castillo, 2007; Botter et al., 2013]. In view of the role of rivers as ecological corridors, conceptual models of species distribution have been developed addressing spatial and temporal biodiversity patterns in riverine systems [Muneepeerakul et al., 2008, 2010]. However, these studies are based on static networks defined solely on the basis of geomorphological features and, therefore, they cannot capture the effect of hydrological processes on the ecological function of rivers. The intertwined link between reach-scale ecological processes and discharge variability in rivers has received much attention in the literature [Camporeale and Ridolfi, 2006; Sabo and Post, 2008; Ceola et al., 2014; Bruno et al., 2015]. For instance, several methods based on the physical simulation of habitats have been developed to predict flow-based alteration of habitat characteristics [Vismara et al., 2001; Parasiewicz and Dunbar, 2001; Vezza et al., 2014]. Nevertheless, the study of the network-scale ecological implications of discharge dynamics is a relatively new discipline [Datry et al., 2014; Jaeger et al., 2014]. Most existing studies are focused on individual river reaches and thereby neglect the spatial dimension of rivers.

According to the river continuum concept, rivers are hydrological continua where ecological processes and species dynamics take place [Vannote et al., 1980]. Following this pioneering concept and its successors [Ward and Stanford, 1983; Junk et al., 1989], the riverscape paradigm offers a new perspective for integrating ecological processes with spatial dynamics of hydrologic regimes [Fausch et al., 2002]. There is growing recognition that river flow regimes control the magnitude of in-stream processes [White et al., 2017], as well as the connectivity between source areas and the catchment outlet, with important implications for biodiversity and ecological functions of rivers across scales [Larned et al., 2010; Datry et al., 2014]. For instance, empirical relationships between fluvial species activities and flow variability have been recognized, in particular for what concerns fish and aquatic invertebrate migration [Monk et al., 2006; Tetzlaff et al., 2008; Belmar et al., 2013; Ceola et al., 2014]. Meanwhile, theoretical approaches have been developed to quantify how the connectivity structure of habitat networks constrains or promotes ecological

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function of rivers [Cote et al., 2009; Eros et al., 2011; Samia et al., 2015]. However, in all these studies a causal connection between river network connectivity and first-order climatic and hydrologic drivers is missing, and little is known about the role of hydrological drivers that shape the ecological function of stream networks.

## Objectives and outline of this thesis

To fill the gap of knowledge about the role of space and time variability of streamflow regimes for river network connectivity, this thesis proposes a probabilistic framework able to investigate quantitatively the principle of causality that drives the link among the following cascade of processes: i) climatic driving forces (rainfall and evapotranspiration), ii) the hydrological response of rivers, iii) the connectivity of the network structure and iv) the fate of ecological species therein. This work is based on a physically based analytical characterization of streamflow regimes at network scale that explicitly accounts for the randomness of rainfall. The flow regime is defined through the probability distribution of discharge, expressed as a function of lumped parameters that embody long-term climatic and landscape features of the contributing areas. Probability distributions of water stages are consequently derived and used to predict the hydrological and ecological impacts of hydro-climatic fluctuations by means of suitable stage and connectivity thresholds.

The remainder of this work is organized as follows:

- Chapter 2 describes how the variability of streamflow regimes and water stage dynamics can be quantified along the river network. A mathematical framework to calculate connectivity measures is also introduced;
- Chapter 3 shows the effects of climate on network-scale connectivity. The spatial variability of hydrologic variables driven by climatic patterns is also presented;
- Chapter 4 investigates the spatial variability of habitats along the network, and it presents the ecological effects of hydrological connectivity alterations;
- Chapter 5 describes the water management policy recently introduced in the

European Union, and it illustrates a new perspective to define ecological flows allowing river connectivity and habitats protection.





# Chapter 2

## Methods

### 2.1 Streamflow model

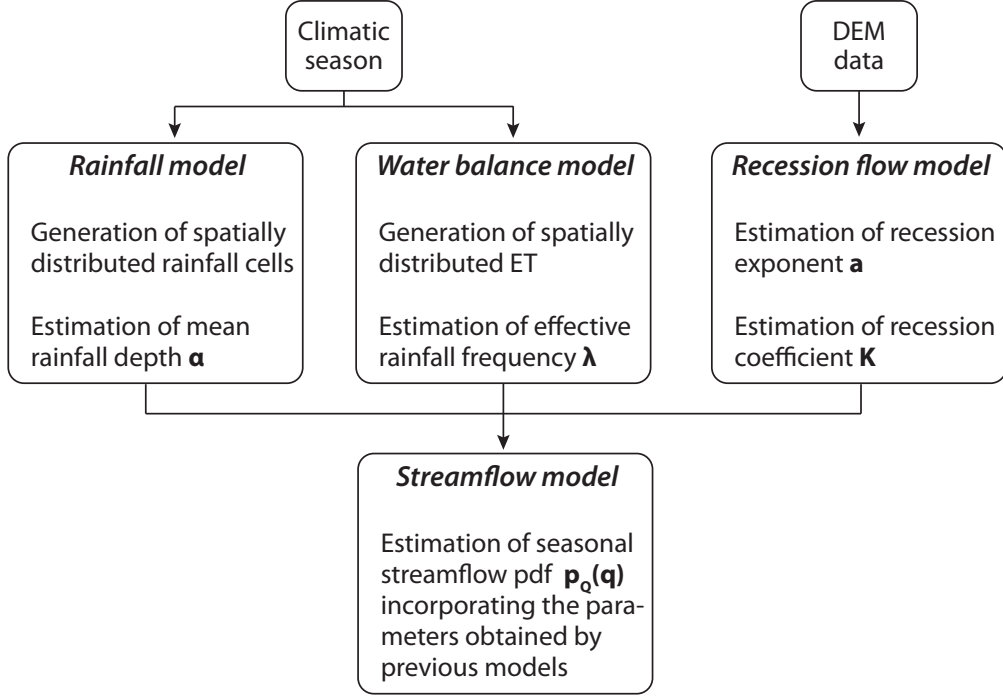
In this thesis, the seasonal probability distribution of streamflows is derived by using a mechanistic analytical model which is based on a stochastic description of catchment-scale water storage dynamics. The model has been introduced by Botter et al. [2007] and further developments have been proposed by Botter et al. [2009] and Doulatyari et al. [2017]. Figure 2.1 outlines the overall modeling approach for estimating parameters of streamflow probability distribution at any site along the river network. The description of the mathematical formulation of the model is reported in the following sections.

#### 2.1.1 Rainfall model

The catchment-scale water storage is controlled by the stochasticity of rainfall. In this work the lumped formulation widely used in literature [Rodriguez-Iturbe et al., 1999; Laio et al., 2001; Ceola et al., 2010; Müller et al., 2014; Park et al., 2014; Basso et al., 2015; Doulatyari et al., 2015; Bertassello et al., 2018] has been extended by implementing a spatio-temporal Poisson process for the stochastic generation of daily rainfall. The occurrence of rain events is described by a counting process  $\{N(t, \mathbb{X}), t \geq 0\}$  of rate  $\lambda_{rain}(t, \mathbb{X}) > 0$  [ $\text{T}^{-1}\text{L}^{-2}$ ], which is a multidimensional

## 2.1. STREAMFLOW MODEL

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**Figure 2.1:** Overall work-flow of the modeling approach. The four main streamflow model parameters are estimated at every point (i.e. every pixel along the river network) as spatially-averaged quantities in the contributing catchment.

Poisson process representing the number of rain events occurring per unit time and per unit area. The process is decoupled into two independent Poisson processes, one in time (with rate  $\lambda_t$  [ $T^{-1}$ ]) and one in a 2D space (with rate  $\lambda_x \lambda_y$  [ $L^{-2}$ ]), according to the equation:

$$\lambda_{rain}(t, \mathbb{X}) = \lambda_t(t) \lambda_x(\mathbb{X}) \lambda_y(\mathbb{X}). \quad (2.1)$$

The marginal time-frequency of the process is obtained via a spatial integration of equation (2.1), thereby incorporating information regarding the spatial density of the events:

$$\begin{aligned} \lambda_{rain}(t) &= \int_{\Omega} \lambda_t(t) \lambda_x(\mathbb{X}) \lambda_y(\mathbb{X}) d\mathbb{X} \\ &= \lambda_t(t) N(\Omega) \end{aligned} \quad (2.2)$$

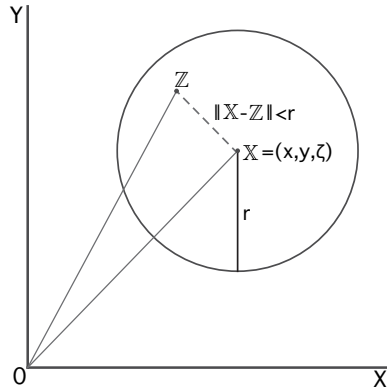
where  $N(\Omega)$  is the number of points (i.e. events) in an arbitrary 2D domain ( $\Omega$ ) contained in the study area. In order to define the position of the rain events within the domain, the coordinates of the rain cells are assumed randomly located in space according to the rate  $\lambda_x=\lambda_y=\text{const}$ . The model reproduces a precipitation process in which rain events are made up by circular cells of constant duration (1 day) during which random rain depths occur. The radius of the cells is assumed as an exponentially distributed independent random variable with the following probability density function (pdf):

$$p_r(r) = \kappa \exp^{-\kappa r}, \quad (2.3)$$

where the parameter  $\kappa = 1/\langle r \rangle$  is the inverse of the mean radius. Likewise, rainfall depth of each cell is an exponentially distributed independent random variable, whose pdf is:

$$p_\zeta(\zeta) = \mu \exp^{-\mu \zeta}, \quad (2.4)$$

where the rate  $\mu = 1/\langle \zeta(\mathbb{X}) \rangle$  is the inverse of the mean rain depth pertaining to each cell centred in  $\mathbb{X} = (x, y)$ . Each rain cell has an independent random rainfall depth that remains constant over the entire duration of the cell. Rainfall depth is assigned to the domain's points on the basis of the cell size. The amount of rainfall is assumed to be uniform within the cell (Figure 2.2).



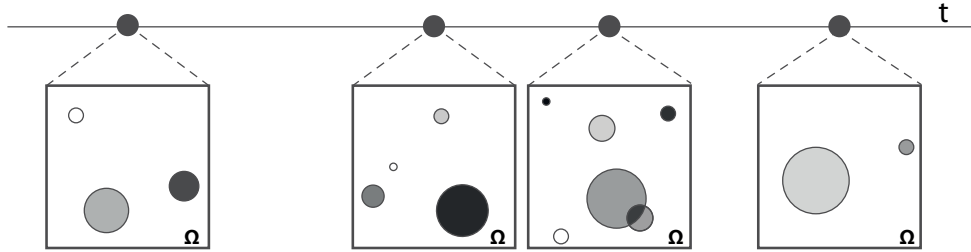
**Figure 2.2:** Rainfall depth is uniform within the cell of given radius  $r$ .

## 2.1. STREAMFLOW MODEL

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The rainfall depth in a generic point  $\mathbb{Z}$  at a given time depends on the location of  $\mathbb{Z}$  in relation to the rain cell: if the point  $\mathbb{Z}$  is inside the cell of radius  $r$  (i.e.  $\|\mathbb{X} - \mathbb{Z}\| < r$ ) the depth assigned to  $\mathbb{Z}$  is the same of the center of the cell, otherwise the depth is zero. It is worth to note that the size of each cell follows the exponential distribution of equation (2.3) and does not depend on the position of the cell in the domain.

Cells can overlap and the rainfall depth is the sum of the rainfall amount of all cells active at given time. Figure 2.3 shows graphically the rainfall process framework.



**Figure 2.3:** Spatio-temporal point process. Rainfall events are exponentially distributed in time  $t$ . Rainfall cells are uniformly distributed in space  $\Omega$ . Cells size and rainfall amount are exponentially distributed independent random variables.

Using such rainfall generation model, spatially-distributed rainfall scenarios are derived to simulate different types of climate (i.e. dry, intermediate and wet [Whittaker, 1975]) and therefore estimate different spatially-averaged rainfall depths ( $\alpha$  [L]) and average rainfall frequency ( $\lambda_P$  [ $T^{-1}$ ]) along the river network. Further details on the simulation set-up are described in Section 2.3.

### 2.1.2 Water balance model

The dynamics of specific streamflow at each node of the network is impacted by positive increments corresponding to rainfall events filling the soil water deficit caused by plant transpiration in the contributing catchment and producing drainage. When the rainfall infiltrating into the hydrologically active layer (i.e. the layer of

soil that actively contributes to the hydrological response, whose porosity and depth are indicated as  $n$  and  $Z_r$ , respectively) exceeds the critical saturation value  $s_1$  (representing the water holding capacity), the excess of water becomes streamflows. Notice that in between rainfall events (whose frequency is  $\lambda_P [T^{-1}]$ ) evapotranspiration, ET [ $LT^{-1}$ ], reduces the soil moisture to the wilting point  $s_w$  (for which ET = 0); hence, the maximum soil water storage capacity available to plants is  $w_0 = (s_1 - s_w)n Z_r$ . Flow-producing rainfall events result from the buffering effect operated by catchments during wetting-drying cycles and they are approximated by a new marked Poisson process, whose frequency is  $\lambda < \lambda_P$ . The ratio  $\phi = \lambda/\lambda_P$  identifies the runoff coefficient (mean discharge scaled to the mean precipitation), which defines the partition of the incoming rainfall into streamflows and ET.  $\phi$  is influenced by climate, soil and vegetation features according to the following equation [Porporato et al., 2004; Botter et al., 2007; Doulatyari et al., 2015]:

$$\phi = \frac{D_I \gamma^{\frac{\gamma}{D_I}} e^{-\gamma}}{\gamma \Gamma(\gamma/D_I, \gamma)}, \quad (2.5)$$

where  $\Gamma(\cdot, \cdot)$  is the lower incomplete Gamma function,  $D_I$  is the Budyko's dryness index (the ratio between the mean potential evapotranspiration  $\langle PET \rangle$  and the mean rainfall  $\langle P \rangle$ ), and  $\gamma$  is the maximum soil water storage capacity  $w_0$ , normalized to the mean rainfall depth in the contributing catchment,  $\alpha$ .

### 2.1.3 Recession flow model

Excess rainfall (fraction of water storage exceeding  $s_1$ ) is released from the soil as river streamflow following a nonlinear catchment-scale storage discharge relation (i.e.  $Q \propto V^\beta$ ) [Kirchner, 2009; Botter et al., 2009]. The resulting dynamic of daily specific discharge (i.e. per unit catchment area) at-a-station is governed by the following equation:

$$\frac{dq(t)}{dt} = -Kq(t)^a + \xi_t, \quad (2.6)$$

where  $K$  and  $a$  are the recession coefficient and the recession exponent, respectively, and  $\xi_t$  formally embeds the stochastic increments of  $q$  induced by effec-

tive rainfall pulses. A geomorphologic recession flow model is then used to estimate the parameter describing the recession flow behaviour ( $a$ ) resulting from the drainage of the contributing catchment [Biswal and Marani, 2010]. The procedure is grounded on the idea that the hydrological response is linked to the morphological properties of the hillslope-network system. Accordingly, the fraction of the network that actively contributes to the flow is assumed to vary over time in relation to the source links retirement ( $dq/dt$  is proportional to the number of the active source links) in the network. Therefore, in this model the recession rate is directly proportional to the distance of the farthest source from the outlet and the recession flow is controlled by the shrinking of the active drainage network. Under certain assumptions (see Biswal and Marani [2010]), the power law exponent of the recession relation (2.6) is equivalent to the exponent of the geomorphologic relation  $N(d)$  vs.  $L(d)$ , where  $N(d)$  is the number of channel sources at a distance  $d$  from the farthest channel heads and  $L(d)$  is the total length of the active drainage network. These functions are derived from morphological data and the scaling exponent (parameter  $a$ ) is estimated through a least-squared regression of the functions  $N$  and  $L$ . It is worth to note that specific morphological requirements are needed to estimate the parameter  $a$  along the network (e.g. at least five junctions are required upstream to each considered point). For further details the reader is directed to Biswal and Marani [2010] and Doulatyari et al. [2015].

The recession coefficient  $K$ , which depends on both the network morphology and the moisture of the catchment, is calculated as  $K = \theta(\alpha \lambda)^{1-a}$  [Doulatyari et al., 2015] where  $(\alpha \lambda)$  is the mean specific discharge,  $a$  is the geomorphic recession exponent and  $\theta$  is the shrinking rate of the network in between rain events.

### 2.1.4 Probabilistic description of streamflows

The streamflow probability density function at network scale emerges directly as a result of aggregation of spatial heterogeneity of climatic and geomorphic features in the contributing areas of each channel site. This is captured by the proposed model by calculating the parameters expressing the intensity and frequency of effective rain events ( $\lambda$  and  $\alpha$ ) and the recession behaviour ( $K$  and  $a$ ) for every point along

the network, as spatially integrated quantities in the corresponding contributing catchment. There are three different types of probability density functions (pdfs) of streamflow, depending on the value of the exponent  $a$  which determines the rate of decrease of  $q$  during recessions. The case  $a = 1$  implies a linear storage-discharge dynamic ( $dq/dt = -kq + \xi_t$ ) in which the decay of flow between subsequent events is exponential-like. The master equation associated with the runoff probability distribution of the specific streamflow per unit catchment area,  $q$ , at time  $t$  reads

$$\begin{aligned} \frac{\partial p(q, t)}{\partial t} &= \frac{\partial [k q p(q, t)]}{\partial q} - \lambda p(q, t) \\ &+ \frac{\lambda}{\alpha k} \int_0^q p(q - z, t) e^{-\frac{z}{\alpha k}} dz, \end{aligned} \quad (2.7)$$

The corresponding steady-state pdf of specific river discharge developed by Botter et al. [2007] is shown here in nondimensional form:

$$p_Q(q) = \frac{\Gamma(\lambda/k)^{-1}}{\alpha k} \left(\frac{q}{\alpha k}\right)^{\frac{\lambda}{k}-1} \exp\left(-\frac{q}{\alpha k}\right). \quad (2.8)$$

Equation (2.8) represents a Gamma distribution with shape parameter  $\lambda/k$  and rate parameter  $\alpha k$ .

The general solution of the probability distribution function for the case  $a \neq 1, 2$  is

$$p_Q(q) = C^* q^{-a} \exp\left[-\frac{q^{2-a}}{\alpha K (2-a)} + \frac{\lambda q^{1-a}}{K (1-a)}\right], \quad (2.9)$$

where  $C^*$  is the normalization constant, such that  $\int_0^\infty p_Q(q) dq = 1$ . Note that, when  $a < 1$  (i.e. the decay of discharge between two subsequent runoff events is more rapid than an exponential-like function) there is an atom of probability in  $q = 0$  (equation (2.10)) that must be added to the continuous part of equation (2.9), as the system tends to remain in a zero discharge state for some time before experiencing a new jump [Botter et al., 2009].

$$p_o = C^* \frac{K}{\lambda} \delta(q). \quad (2.10)$$

## 2.1. STREAMFLOW MODEL

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Moreover, the extension of the equation (2.9) to the case  $a = 2$  is an inverse Gamma [Deal et al., 2018]:

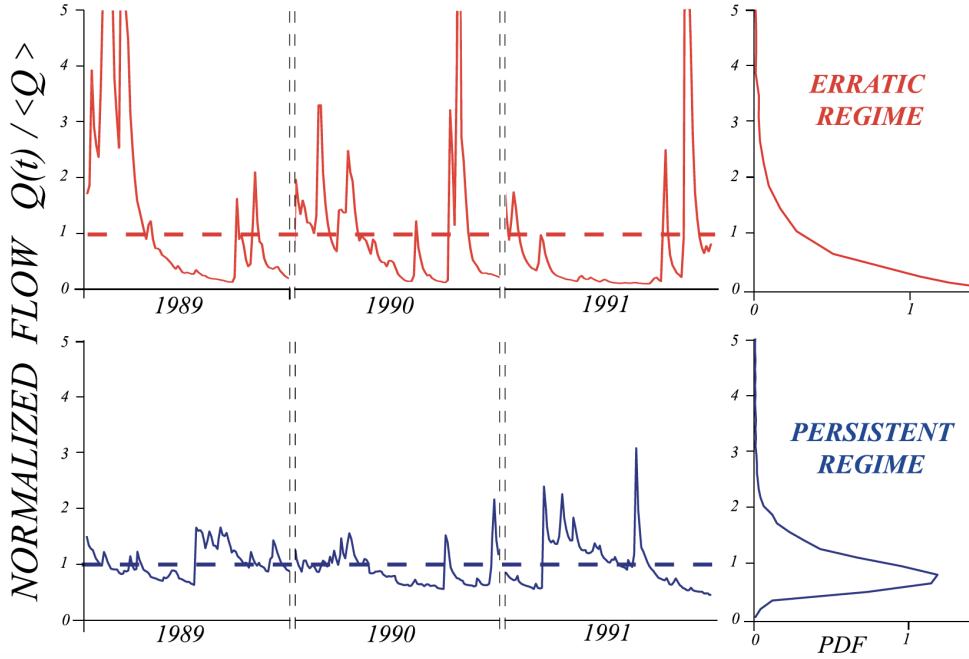
$$p(q) = \left[ \frac{\left(\frac{\lambda}{K}\right)^{\frac{1}{\alpha K} + 1}}{\Gamma\left(\frac{1}{\alpha K} + 1\right)} \right] q^{-2 - \frac{1}{\alpha K}} e^{-\frac{\lambda}{Kq}}, \quad (2.11)$$

where the first term on the right-hand side expresses the normalization constant in which  $\Gamma(\cdot)$  is the Gamma function with argument  $(\cdot)$ .

All these equations were applied pointwise along all the streams of the test catchment. Therefore, model parameters are the expression of climatic and landscape attributes in the contributing catchment and vary in space along the network.

In this framework flow regimes can be classified based on the variability of river flows [Botter et al., 2013], which results from the interplay between the frequency of flow-producing rainfall events and the mean catchment response time. When the mean inter-arrival of effective rainfall events is shorter than the duration of the flow pulses delivered from the contributing catchment, a persistent supply of water is guaranteed to the stream from catchment soils. This type of regime is termed persistent as the coefficient of variability of streamflow ( $CV_Q$ ) is smaller than 1 (Figure 2.4). On the contrary, when the mean inter-arrival between flow-producing rainfall events is larger than the typical duration of the resulting flow pulses, significant streamflow fluctuations are observed. In this case the preferential state of the system is typically lower than the mean and the flow regime (termed erratic) is characterized by a pronounced flow variability ( $CV_Q > 1$ ).





**Figure 2.4:** Typical behaviour of river flow dynamics in erratic and persistent regimes.

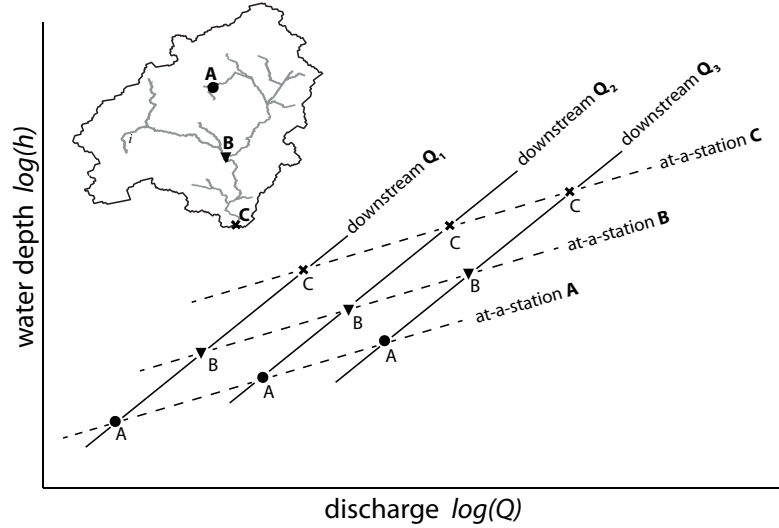
## 2.2 Stage dynamics and connectivity metrics

The temporal and spatial variability of streamflows affects patterns of hydraulic variables (e.g. water depth, flow velocity and bottom shear stress), which influence communities distribution and species abundance in fluvial ecosystems [Ceola et al., 2014]. In this framework river width, depth and velocity are assumed to increase downstream according to the power-law relationship proposed by [Leopold and Maddock, 1953]. Accordingly, site-specific probability distribution functions of relevant flow variables can be derived from the corresponding streamflow distribution,  $p_Q(q)$ , using additional information on the geomorphic and hydraulic properties of the river cross section. In this framework water depth (i.e. stage) is assumed to scale with discharge as [Leopold and Maddock, 1953]:

$$h = h_0 Q^\delta = h_0 (Aq)^\delta, \quad (2.12)$$

## 2.2. STAGE DYNAMICS AND CONNECTIVITY METRICS

where  $A$  is the catchment area,  $h_0$  is the stage associated to the formative discharge and  $\delta$  is a dimensionless parameter experimentally found to be close to 0.3 for many rivers worldwide [Raymond et al., 2012; Ceola et al., 2014]. Notwithstanding that equation (2.12) refers to the “at-a-station” stage-discharge relationship, the parameter  $h_0$ , which depends on the geometrical characteristic of the cross section, is assumed equal to  $0.04 [T^\delta L^{1-3\delta}]$  in this study. For a given basin (Figure 2.5), the at-a-station storage-discharge relationship assumes that the parameter  $h_0$  scales downstream (i.e.  $h = h_0(i)Q^{\delta_{at-a-station}}$ , where  $i$  is the considered node); the downstream relationship, on the other hand, assumes that  $h_0$  is uniform downstream for a given frequency of discharge (i.e.  $h = h_0Q^{\delta_{downstream}}$ ). Since the slope  $\delta_{at-a-station} \simeq \delta_{downstream}$ , as demonstrated by Leopold and Maddock [1953], hence  $h_0(i) = h_0 = const$  along the network.



**Figure 2.5:** Relation of depth to discharge, for selected cross sections (dashed lines) and in downstream direction for given discharge frequencies (solid lines).

By coupling equations (2.9) and (2.12), the following analytical expression of the stage probability distribution function is obtained:

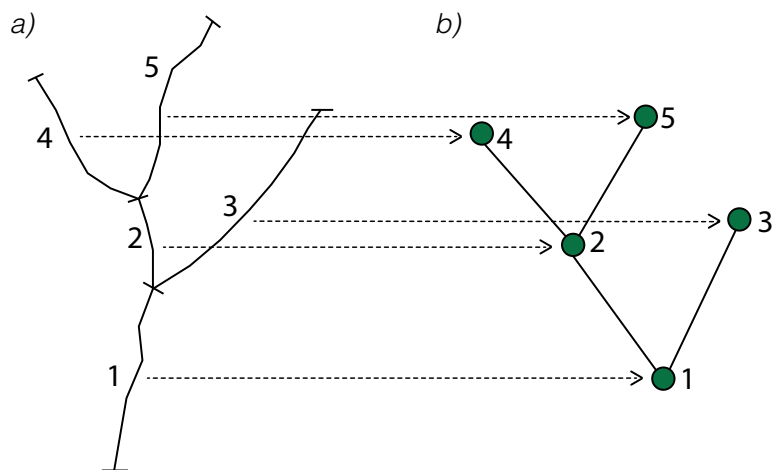
$$\begin{aligned}
p_H(h) \propto & \frac{(h/h_o)^{\frac{(1-a)}{\delta}}}{\delta h A^{(1-a)}} \exp \left[ -\frac{1}{\alpha K (2-a)} \left( \frac{(h/h_o)^{1/\delta}}{A} \right)^{(2-a)} + \right. \\
& \left. + \frac{\lambda}{K(1-a)} \left( \frac{(h/h_o)^{1/\delta}}{A} \right)^{(1-a)} \right].
\end{aligned} \tag{2.13}$$

Water stage is a major control on the physical connection between two nodes of a river network. For instance, large fish migrating towards headwater streams during drought periods may find it difficult to reach their target in case where minimum flow requirements are not guaranteed [Tetzlaff et al., 2005; Lazzaro et al., 2017]. Likewise, many ecological species could be particularly vulnerable to predation during migration in shallow water [Jonsson et al., 2007]. Hence, low stages (associated to low flows) can be seen as a physical barrier that decreases the chances of completing migratory movements, with implications for the composition of structured metacommunities [Campbell et al., 2015]. In line with Lazzaro et al. [2017], this thesis assumes here the existence of a minimum threshold stage,  $h^*$ , which is necessary to trigger the movement of biological species. When  $h < h^*$ , the corresponding stream is assumed to be too dry to maintain the connection between upstream and downstream sites. In general,  $h^*$  is a function of the specific species considered and its sensitivity to droughts. For instance, large fishes are likely to be characterized by larger values of  $h^*$  compared to bacteria and propagules.

Considering the connectivity as a categorical and instantaneous variable (connected versus disconnected) is less informative than focusing on temporally integrated quantities such as the frequency and duration of hydrological conditions that allow for species migration ( $h \geq h^*$ ). Therefore, the connectivity of a given reach is evaluated by considering the fraction of days within a season during which hydrological conditions favourable to species movement are observed. The latter is calculated as the exceedance probability of the stage threshold  $h^*$ . This probability represents the probability of experiencing water stages that ensure a physical connection between different sites of the network.

In this thesis, connectivity metrics are introduced to evaluate the overall connectivity of the network. They are based on the mathematical structure of the graph

theory (Figure 2.6). In particular, branches of the river are seen as nodes, whereas confluences are represented by links.



**Figure 2.6:** a) A river network with five reaches. b) Spatial graph representation of the river network.

The relational schema of the graph is undirected, assuming that ecological communities in riverine landscapes can move either upstream or downstream. In this work different connectivity measures are used:

- Local connectivity:

$$C_{local(n)} = \int_{h^*}^{+\infty} p_H(h)_{(n)} dh, \quad (2.14)$$

where  $p_H(h)_{(n)}$  is the stage density function for the  $n$ -th node.  $C_{local}$  measures the passage probability through the node ( $n$ ).

- Path connectivity:

$$C_{path(j,k)} = \prod_{n \in j \rightarrow k} C_{local(n)}, \quad (2.15)$$

where  $n$  includes the set of nodes belonging to the path that connects node  $j$  to  $k$ .  $C_{path}$  is calculated as the product of the  $C_{local}$  of all nodes from  $j$  to  $k$  and expresses the connection probability between pair of nodes, according to the hydrological dynamics in all the reaches of the path connecting  $j$  and  $k$ . In the calculation of  $C_{path}$ , temporal correlation of flows are neglected.

- Node connectivity:

$$C_{node(k)} = \frac{1}{N-1} \sum_{\substack{j=1 \\ k \neq j}}^N C_{path(j,k)}, \quad (2.16)$$

where  $N$  is the total number of nodes and  $j$  and  $k$  are generic nodes of the network.  $C_{node}$  is calculated as the average value of the connectivity of the paths directed to the node  $k$ . Accordingly, it expresses the probability for a single node to be connected with all the other nodes of the network.

- Network connectivity:

$$C_{ntw} = \frac{1}{N} \sum_{n=1}^N C_{node(n)}. \quad (2.17)$$

$C_{ntw}$  expresses the connection probability of all the possible pairs of nodes within the entire river network;  $C_{ntw}$  is the average value of the probability to connect any site to all other sites in the network.

### 2.3 Simulation set-up

This section describes the estimation of the flow model parameters and the simulation set up to predict spatial patterns of the hydrological connectivity in a theoretical case study. At first, rain cells are generated using the 3-dimensional Poisson rainfall model. An example of the output is reported in Figures 2.7. Table 2.1 shows parameters ranges used to reproduce different climatic scenarios. Notice that, in the proposed framework a periodic space domain is used to reduce bias

### 2.3. SIMULATION SET-UP

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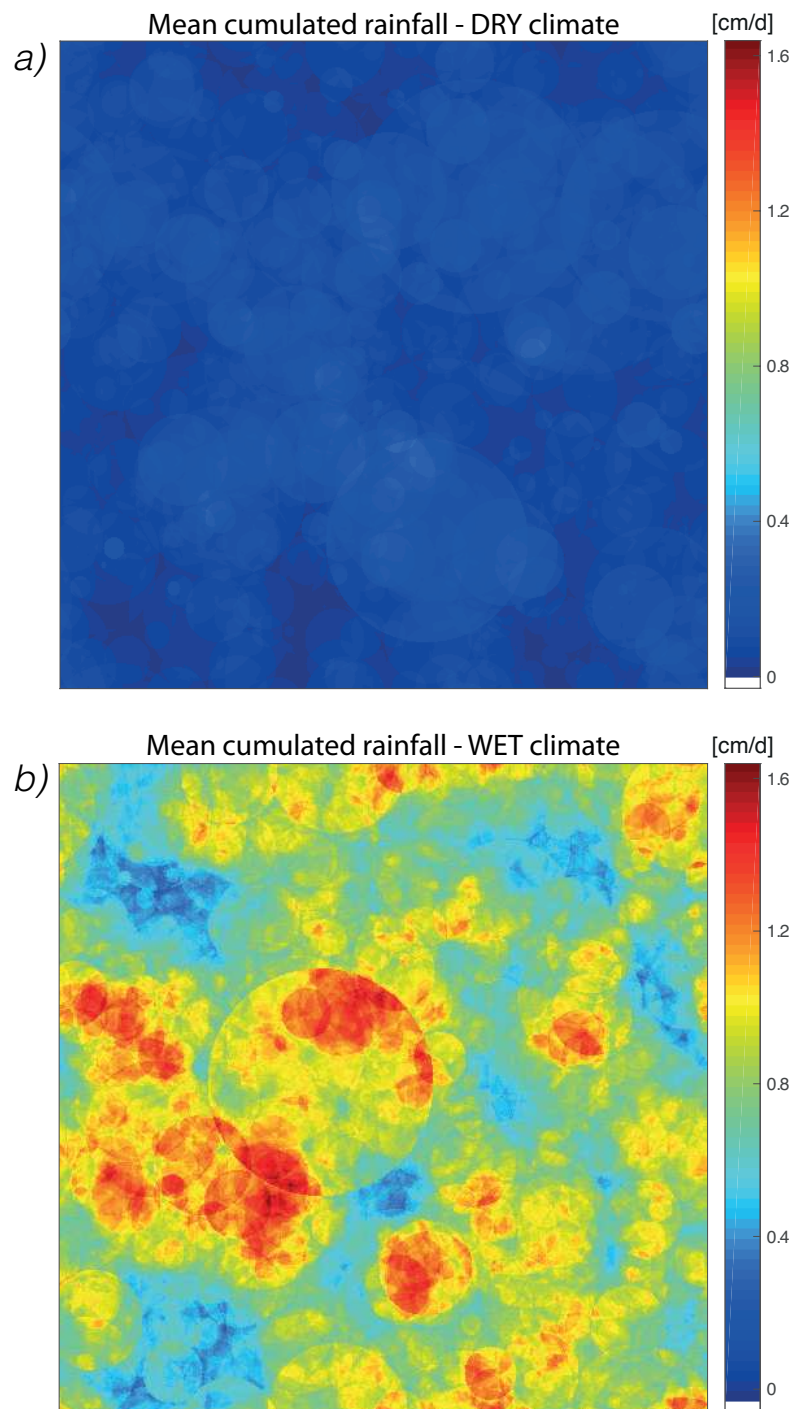
from edge effects. Additionally, the integral scale of the generated rainfall fields is smaller than the linear scale of the space domain.

**Table 2.1:** Model parameters for rainfall generation model

Parameter	Symbol	Value	Units
Temporal frequency	$\lambda_t$	$1.5 \div 11$	[1/d]
Spatial frequency along x direction	$\lambda_x$	$0.7 \div 1.2$	[1/m]
Spatial frequency along y direction	$\lambda_y$	$0.7 \div 1.2$	[1/m]
Mean rain cell depth	$\langle \zeta \rangle$	$15 \div 4500$	[mm]
Mean rain cell radius	$\langle r \rangle$	$1 \div 2$	[km]

The average frequency of rainfall events along the network,  $\lambda_P$ , is estimated by evaluating the fraction of rainy days (rain depth larger than 1 mm) during the considered time period (e.g. a season of 100 days) occurring in the upstream contributing area of each node. The mean rainfall depth,  $\alpha$ , is calculated for each realization of the rainfall model as the spatially-averaged rain intensity during wet days (i.e., the intensity of rainfall is divided by the number of wet days and the catchment area drained by each node).

The frequency of the effective rainfall,  $\lambda$ , is estimated considering the censoring effect of the soil catchment that results in the reduction of flow-producing rate controlled by evapotranspiration. To this aim, spatially distributed values of potential evapotranspiration (i.e.  $PET(\mathbb{X})$ ) are used. The runoff coefficient  $\phi$  is estimated using equation (2.5) considering the mean potential evapotranspiration as a spatially-averaged value in the upstream contributing catchment. Table 2.2 shows literature values of porosity  $n$ , rooting depth  $Z_r$ , soil moisture at saturation  $s_1$  and wilting point  $s_w$ , that are considered and incorporated in equation (2.5).  $\lambda$  is then calculated as the product  $\phi \lambda_P$  for every node of the network.



**Figure 2.7:** Simulation of rainfall scenarios: a) Dry climate; b) Wet climate

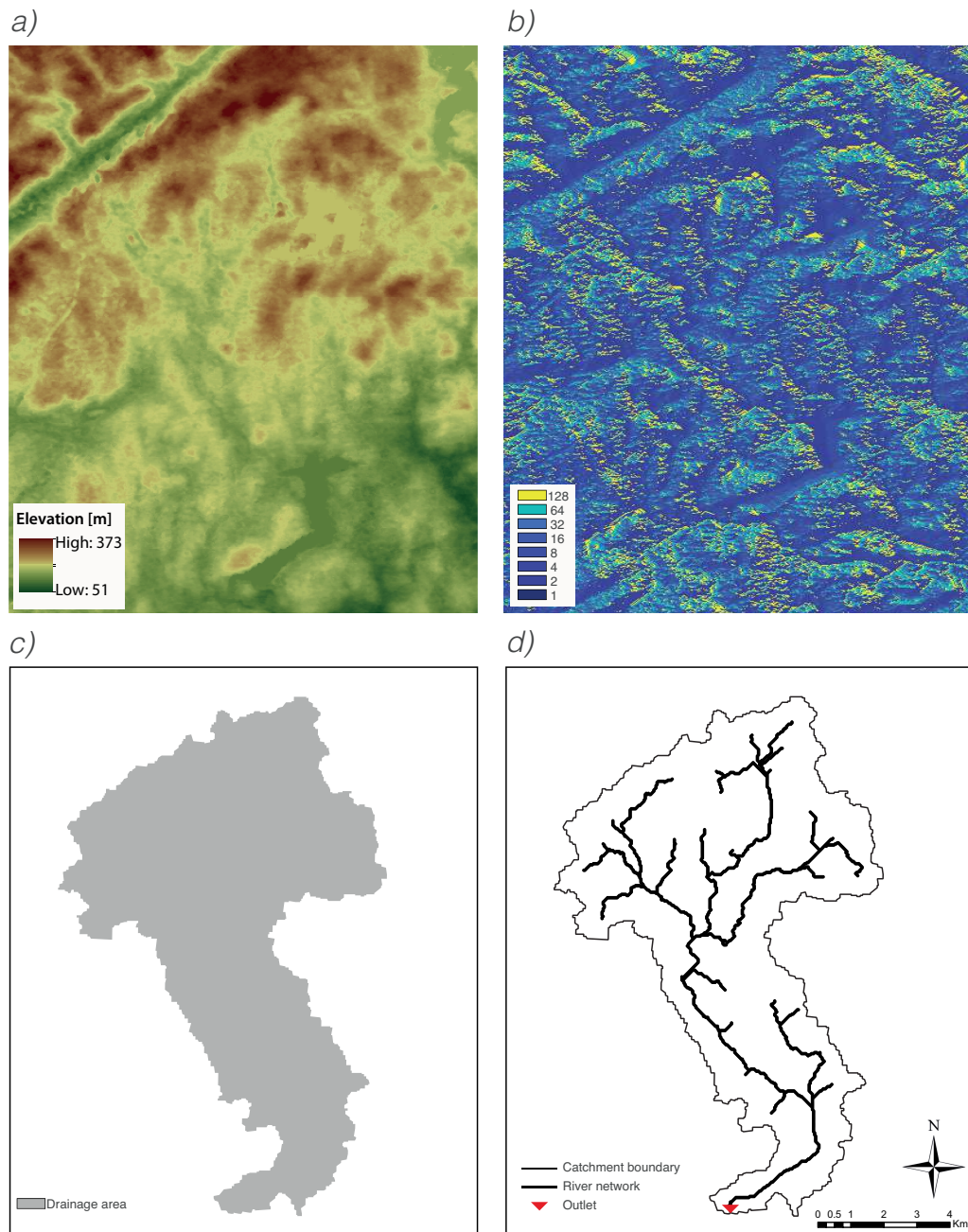
**Table 2.2:** Water balance parameters

Parameter	Symbol	Value	Units
Soil moisture at saturation	$s_1$	0.5	[-]
Wilting point	$s_w$	0.2	[-]
Porosity	$n$	0.3	[-]
Rooting depth	$Z_r$	130	[mm]

Parameters defining the storage-discharge relations are then derived by river network analysis as detailed in Section 2.1.3. The river network estimation is derived from a representative DEM from which 8-flow direction and flow accumulation maps are calculated to identify channel network, as shown in Figure 2.8. The DEM analysis and the network structure definition are used to estimate the at-a-point recession parameter  $a$ . Subsequently, considering both climatic and geomorphic features of the catchment, the recession constant  $K$  is calculated as  $K = \theta(\alpha \lambda)^{1-a}$  using a constant value of  $\theta$  equal to  $0.2 \text{ d}^{-1}$  [Doulatyari et al., 2015].

The streamflow distribution calculated pointwise using equation 2.9 is finally used as input to derive the spatial and temporal variability of water depth by applying equation 2.12. The parameters of the nonlinear stage-discharge relation are assumed to be constant along the river network, as discussed in section 2.2. Connectivity metrics are then calculated using two sets of critical stages: i) low stages ( $h^* = 50 \div 150 \text{ mm}$ ), that are assumed to be associated to smaller general species, such as bacteria or microinvertebrates, requiring small amounts of water to disperse; ii) medium to high stages ( $h^* = 200 \div 500 \text{ mm}$ ), that are representative of fish species that need relatively high water stages for migration [Armstrong et al., 2003].





**Figure 2.8:** a) DEM of the representative case study; b) 8-flow direction map; c) Flow accumulation map; d) Channel network. Estimated channel length is 72 km and the catchment area is close to 87 km<sup>2</sup>.



# Chapter 3

## Hydrological Connectivity

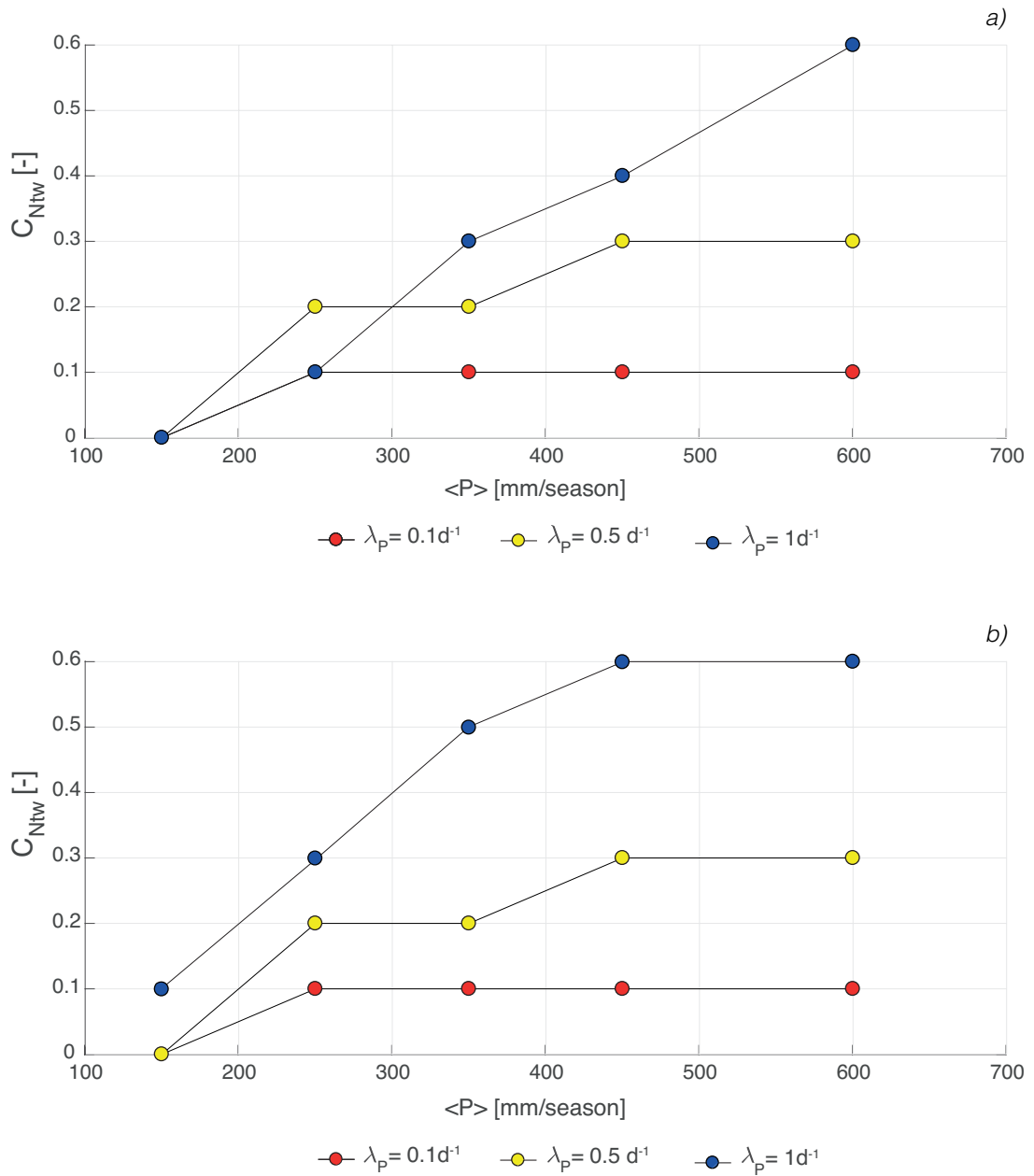
Rainfall frequency, intensity and amount are major drivers of the availability and variability of streamflows, and thus they are likely to impact significantly the hydrological connectivity of rivers. This chapter describes the hydroclimatic effects on both the network-scale connectivity and the spatial variability of hydrological variables.

### 3.1 Effects of climate on network connectivity

In this section, the network connectivity is studied considering three different rainfall frequencies (e.g.  $\lambda_P = 0.1 \text{ d}^{-1}$ ,  $\lambda_P = 0.5 \text{ d}^{-1}$  and  $\lambda_P = 1 \text{ d}^{-1}$ ) under various climatic scenarios in terms of mean precipitation  $\langle P \rangle$  and mean potential evapotranspiration  $\langle \text{PET} \rangle$ .

The network connectivity  $C_{ntw}$  typically increases by increasing the mean precipitation depth, if the frequency of the events is constant (Figure 3.1a). Moreover, in wet climates ( $\langle P \rangle > 300 \text{ mm/season}$ ) connectivity also increases with increasing  $\lambda_P$  when the precipitation amount is kept constant. For high frequencies of rain events the soil moisture is often higher of the field capacity, thereby originating persistent flow regimes ( $CV_Q < 1$ ) with relatively high flows. Conversely, in intermediate climates ( $\langle P \rangle = 200 \div 300 \text{ mm/season}$ )  $C_{ntw}$  can increase also when the rainfall frequency is reduced, as low frequency events have higher intensity (as the

### 3.1. EFFECTS OF CLIMATE ON NETWORK CONNECTIVITY

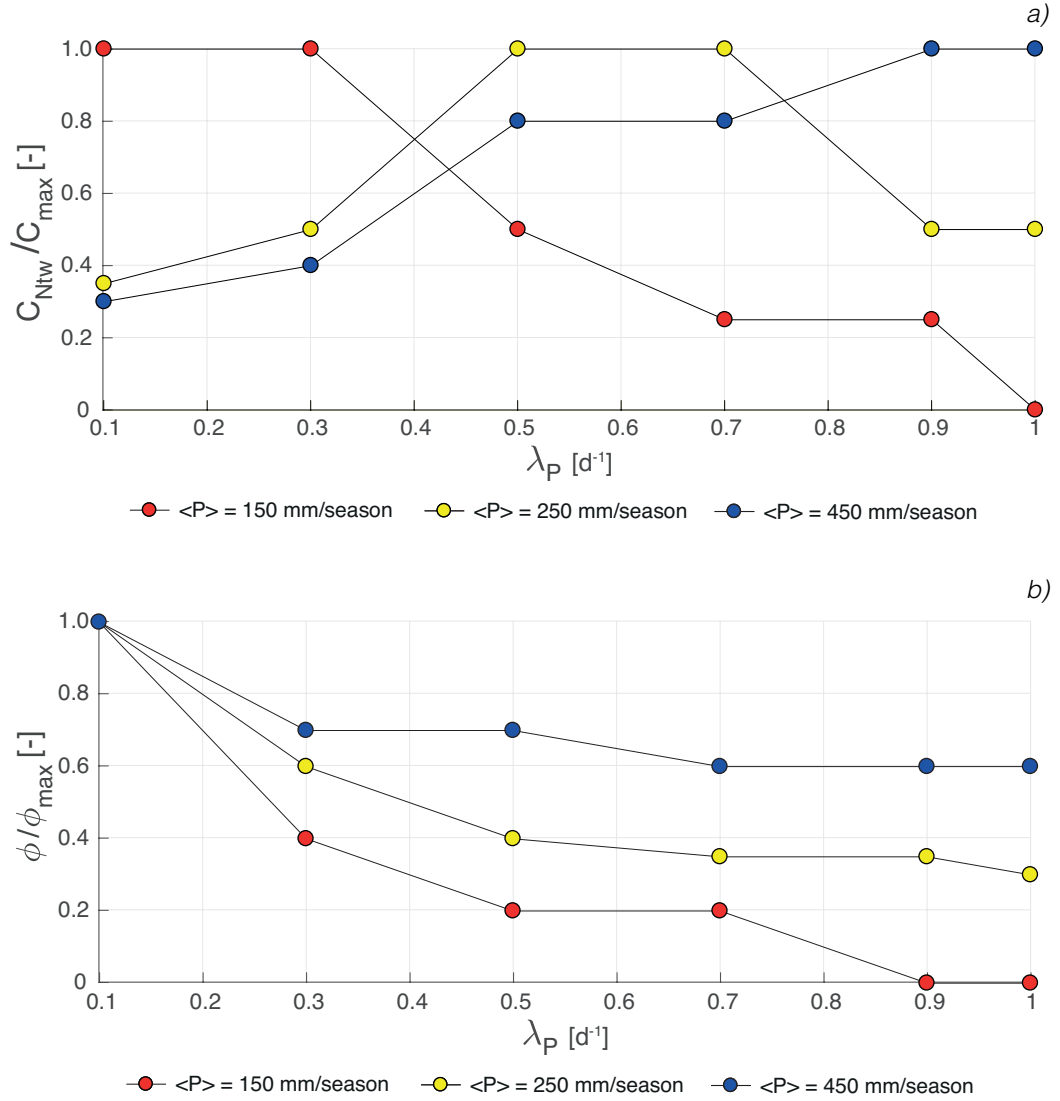


**Figure 3.1:** Network connectivity for increasing precipitation and different rainfall frequencies assuming a)  $\langle \text{PET} \rangle = 3.5 \text{ mm/d}$ ; b)  $\langle \text{PET} \rangle = 0.5 \text{ mm/d}$ .

rainfall amount is constant). This circumstance reduces the buffering capacity of the catchment and increases the mean streamflow and the mean stage along the network. For low values of  $\langle \text{PET} \rangle$  (0.5 mm/d, Figure 3.1b) the connectivity generally increases when: i)  $\lambda_P$  is kept constant and  $\langle P \rangle$  increases; and ii)  $\langle P \rangle$  is kept constant and  $\lambda_P$  increases. The latter mechanism is particularly evident under wet climatic conditions, during which large rainfall inputs inhibit the buffering capacity of the soil leading to higher mean streamflows and higher connectivities throughout the network.

Figure 3.2a shows how the network connectivity changes as a function of rainfall frequency and for increasing values of  $\langle P \rangle$ , combined to a relatively high and uniform  $\langle \text{PET} \rangle$  (3.5 mm/d). In particular, in the wet scenario where  $\langle P \rangle$  is larger than the  $\langle \text{PET} \rangle$ , the connectivity increases with the frequency of rainfall due to the higher mean streamflows associated to larger  $\lambda_P$ . This prevents significant soil water deficits in between events, as confirmed by high values of the runoff coefficient in this case (blue dots in Figure 3.2b), and leads to persistent hump-shaped flow regimes, especially in the downstream reaches of the network. Conversely, when evapotranspiration is higher than the mean rainfall, the runoff coefficient decreases as the rainfall frequency increases. This suggests that in dry scenarios streamflow regimes could be erratic throughout the river network, with enhanced network fragmentation for larger rainfall frequencies. When  $\langle \text{PET} \rangle$  is reduced to 0.5 mm/d the connectivity systematically increases for higher rainfall frequencies, regardless of the underlying precipitation amount. In these circumstances, the runoff coefficient  $\phi$  slowly decreases with  $\lambda_P$ , though maintaining relatively high values under all climatic scenarios. This typically generates persistent flow regimes, in which the variability of flows decreases as  $\lambda_P$  increases. The non-exceedance probability of the critical stage  $h^*$ ,  $P[h < h^*]$ , is thus reduced and the network connectivity increases.

### 3.1. EFFECTS OF CLIMATE ON NETWORK CONNECTIVITY



**Figure 3.2:** a) Normalized  $C_{Ntw}$  for increasing rainfall frequency with different mean precipitation assuming uniform  $\langle PET \rangle = 3.5$  mm/d; b) normalized  $\phi$  for increasing rainfall frequency with different mean precipitation assuming uniform  $\langle ET \rangle = 3.5$  mm/d;

Further analysis is carried out by evaluating the effect of spatial patterns of evapotranspiration (namely N-S, S-N, E-W and W-E direction) on  $C_{ntw}$  for each climate scenario.  $C_{ntw}$  shows similar values regardless of the dominant direction of evapotranspiration gradients. This means that spatial patterns of PET do not affect the average connectivity at network scale.

## 3.2 Spatial variability of hydrological variables

The analysis of the spatial patterns of hydrological and ecological variables is here performed focusing on a relatively dry climatic setting (i.e.  $\langle P \rangle = 150$  mm/season and spatially uniform  $\langle \text{PET} \rangle = 3.5$  mm/d). Three different values of the rainfall frequency are investigated (e.g.  $\lambda_P = 0.1$  d<sup>-1</sup>,  $\lambda_P = 0.5$  d<sup>-1</sup> and  $\lambda_P = 1$  d<sup>-1</sup>).

Simulations indicate that the runoff coefficient,  $\phi$ , plays a critical role in shaping spatial patterns of connectivity. Under dry climates,  $\phi$  generally exhibits a power-law dependence on the drainage area (i.e.  $\phi \propto A^{-\beta}$ ). This is due to the reduction of the mean rainfall intensity and the increase of precipitation frequency for larger contributing areas (the larger the catchment area, the higher the occurrence probability of local rain events that involve only a small portion of the basin). Low rain frequency ( $\lambda_P = 0.1$  d<sup>-1</sup>) lead to high values of  $\phi$  in downstream sites (Figure 3.3b). Hence, the intensity of the events is sufficient to generate persistent flow regimes in most channel sites, thereby increasing  $C_{ntw}$ . Conversely, frequent events with reduced intensity ( $\lambda_P = 1$  d<sup>-1</sup>) entail rather small and uniform values of  $\phi$  along the network, reducing  $C_{ntw}$  significantly.

The mean water depth,  $\langle h \rangle$ , generally increases with the drainage area, as a byproduct of the scaling relation (equation (2.12)). However,  $\langle h \rangle$  slightly increases with  $A$  when  $\lambda_P = 1$  d<sup>-1</sup> as long as  $\phi$  decreases with  $A$  as a power-law with an exponent  $\beta$  close to 1 (Figure 3.4). Therefore, when  $\beta$  approaches 1 the mean depth  $\langle h \rangle$ , which scales as  $[A^{1-\beta}]^\delta$ , tends to remain constant throughout the network. When  $\langle h \rangle$  remains nearly uniform along the network, the connectivity is affected by second-order moments of the stage pdf. In particular, for increasing  $\lambda_P$  the coefficient of variation of the stage distribution decreases, and the hydrological

### 3.2. SPATIAL VARIABILITY OF HYDROLOGICAL VARIABLES

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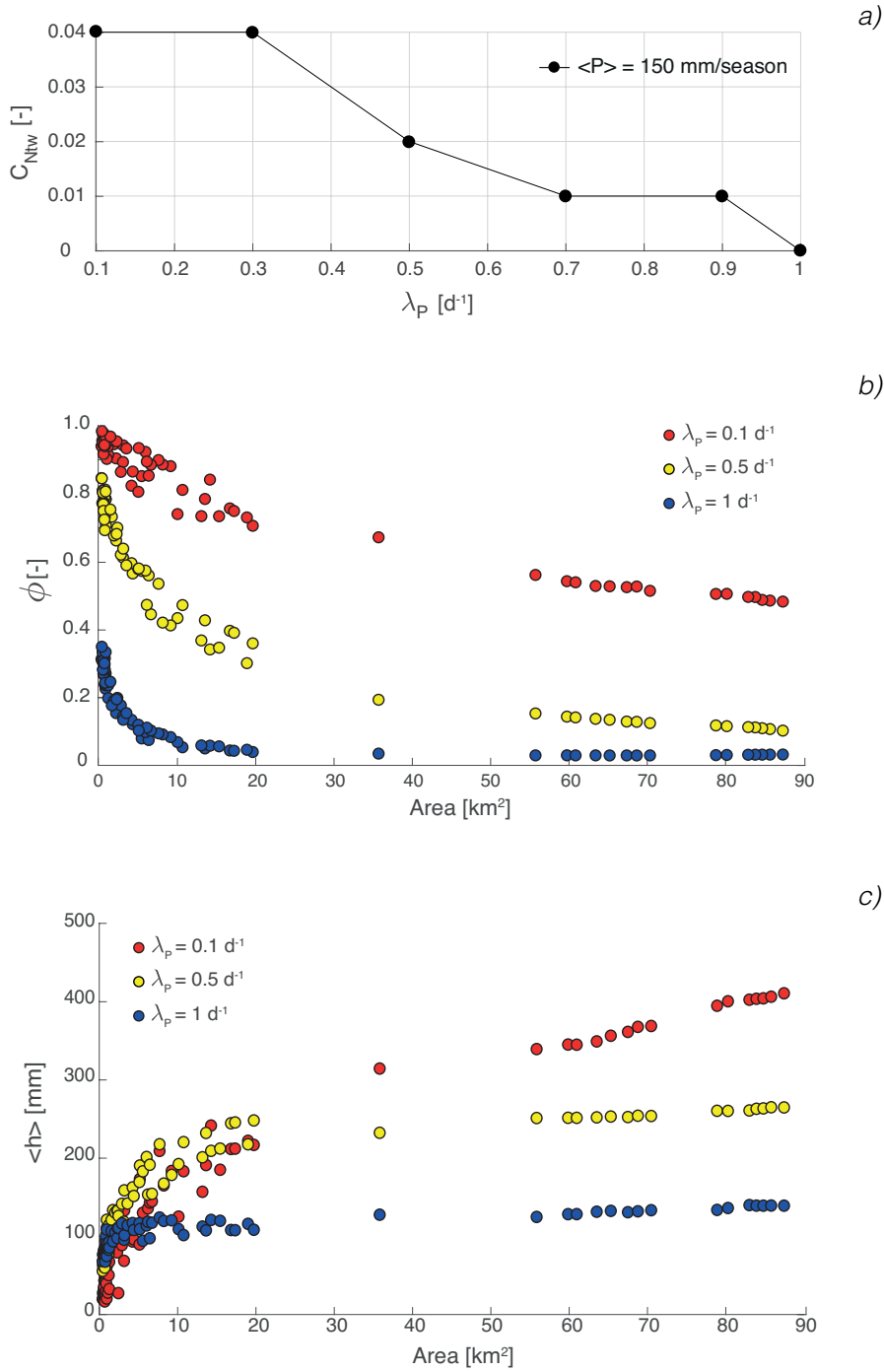
connectivity becomes a function of the relationship between  $\langle h \rangle$  and  $h^*$ .

Figure 3.5 shows the water stage pdfs for increasing frequencies of rainfall events, assuming  $\langle P \rangle = 150$  mm/season. For high rainfall frequency (Figure 3.5c), the pronounced decrease of  $\phi$  with A promotes high probabilities of relatively small water stages ( $\langle h \rangle \simeq 150$  mm) in downstream sites. This strongly reduces the connectivity when  $h^* > 250$  mm. Therefore, in most circumstances, the runoff coefficient represents a key factor governing the spatial patterns of the probability distribution of water stage along the network, and the ensuing connectivity.

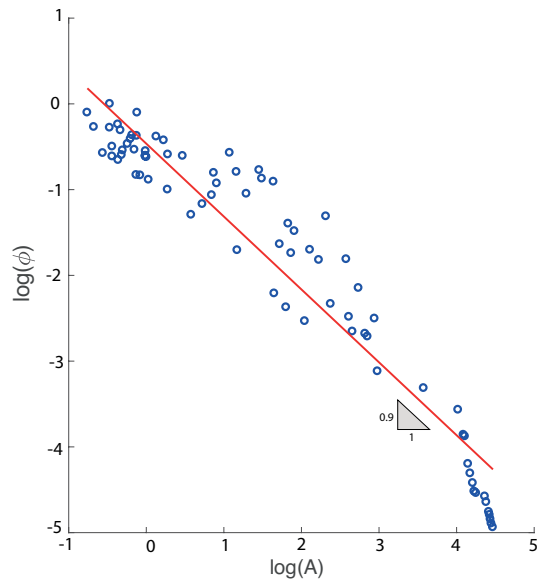
However, hydrological connectivity is also strongly dependent of the stage threshold  $h^*$ . Generally, low thresholds ( $h^* = 50$  mm) produce high connectivity everywhere along the network regardless of  $\lambda_P$ . Higher thresholds ( $h^* = 250$  mm), instead, produce high connectivities in downstream sites ( $C_{local} = 0.7 \div 1$ ) only for low frequency of rainfall ( $\lambda_P = 0.1$  d<sup>-1</sup>). Conversely, very low connectivities are observed throughout the network ( $C_{local} < 0.1$ ) for higher values of  $\lambda_P$  ( $\lambda_P = 1$  d<sup>-1</sup>). Therefore, under the same mean precipitation and for different values of  $\lambda_P$ , different stage thresholds produce heterogeneous patterns of connectivity along the network (Figure 3.6). Although the spatial variability of evapotranspiration does not affect the average connectivity of the network, the impact of PET patterns on stage pdfs and local connectivity in dry climatic conditions is noticeable. Figure 3.7 shows the probability distributions of water stages along the network for three different evapotranspiration patterns. When  $\langle \text{PET} \rangle$  is assumed to be spatially uniform,  $\langle h \rangle$  increases in downstream sites as driven by the increase of the drainage area. If  $\langle \text{PET} \rangle$  is assumed to increase downstream,  $\langle h \rangle$  slightly decreases along the network because the increase of spatially averaged PET from upstream to downstream sites enhances the decrease of the runoff coefficient for increasing contributing areas (Figure 3.7b). The increase of  $\langle h \rangle$  for larger contributing areas becomes less pronounced when  $\langle \text{PET} \rangle$  is assumed to decrease downstream. Spatial patterns of  $\langle \text{PET} \rangle$  affect the tail of  $p_H(h)$ , which is a second-order control on connectivity patterns. The probability of high water stages is reduced when  $\langle \text{PET} \rangle$  is spatially variable (insets of Figure 3.7b and c), with a reduction of connectivity especially in downstream sites. As a consequence, the same stage threshold used



### 3. HYDROLOGICAL CONNECTIVITY



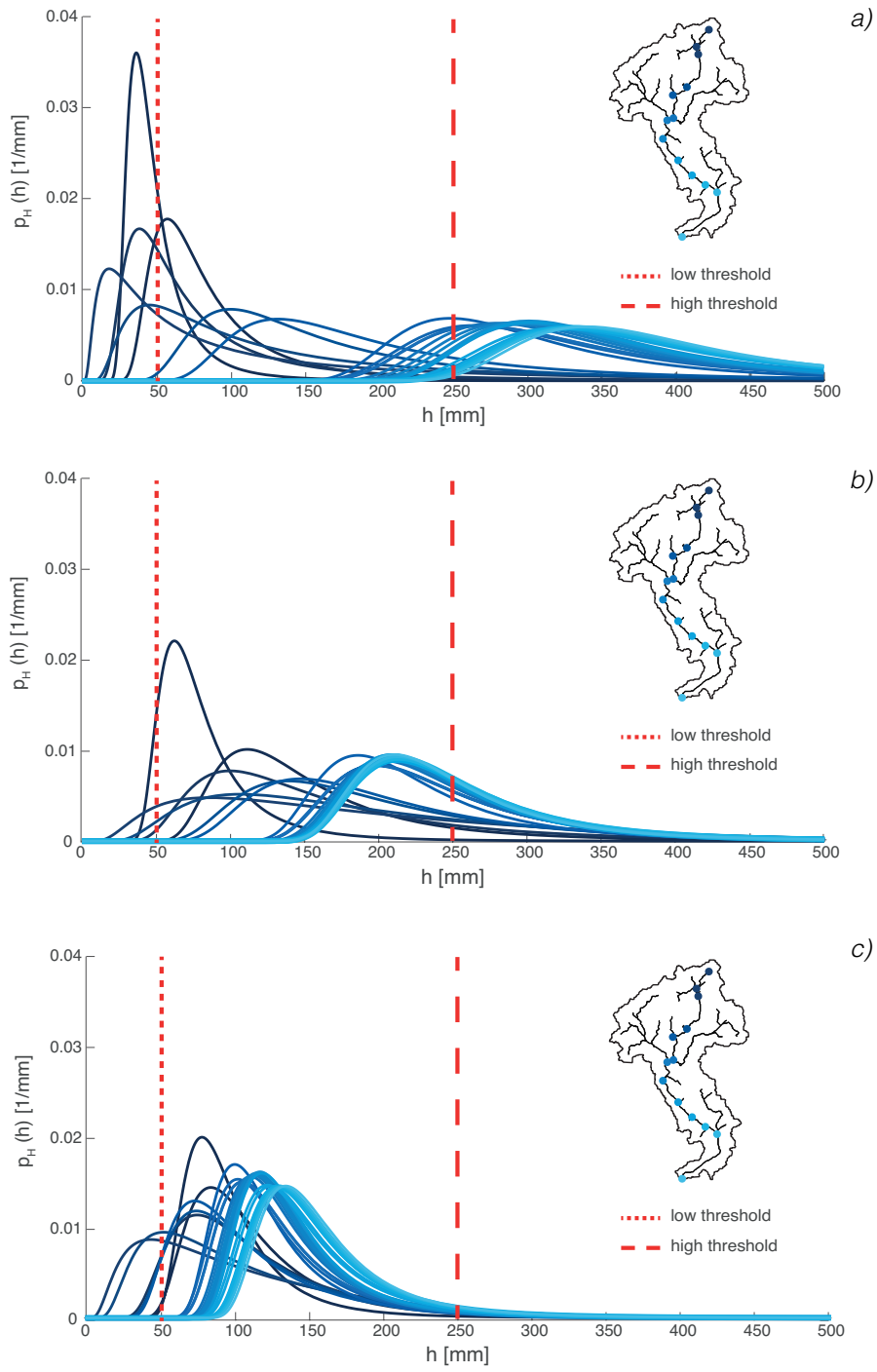
**Figure 3.3:** a) Pattern of  $C_{ntw}$  for increasing rainfall frequency; b) Scaling relation of the runoff coefficient  $\phi$ ; c) Scaling relation of the mean stage  $\langle h \rangle$ . All simulations refer to a dry climatic settings ( $\langle P \rangle = 150$  mm/season and uniform  $\langle PET \rangle = 3.5$  mm/d).



**Figure 3.4:** The scaling exponent  $\beta$  of the runoff coefficient when  $\langle P \rangle = 150$  mm/season and  $\lambda_P = 1$  d<sup>-1</sup> is close to 1.

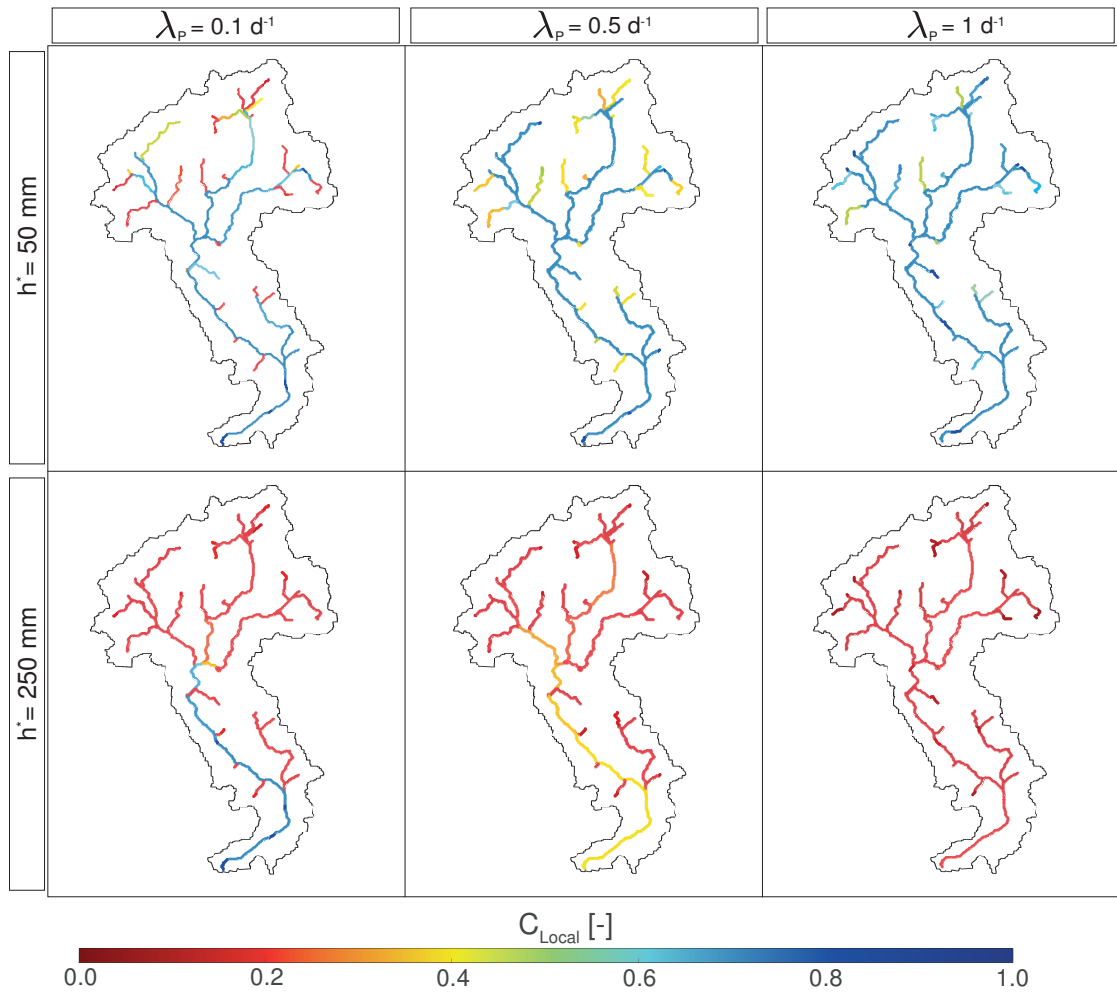
with different PET patterns produces different spatial distributions of hydrological connectivity at local scale (Figure 3.8). Overall, the analysis indicates the emergence of unexpected spatial patterns of connectivity induced by patterns of evapotranspiration, especially under arid climatic conditions. High values of local connectivity are observed not only in downstream sites (where the mean stage is typically higher) but also in river reaches located in the middle of the network (Figure 3.8b and c).

### 3. HYDROLOGICAL CONNECTIVITY



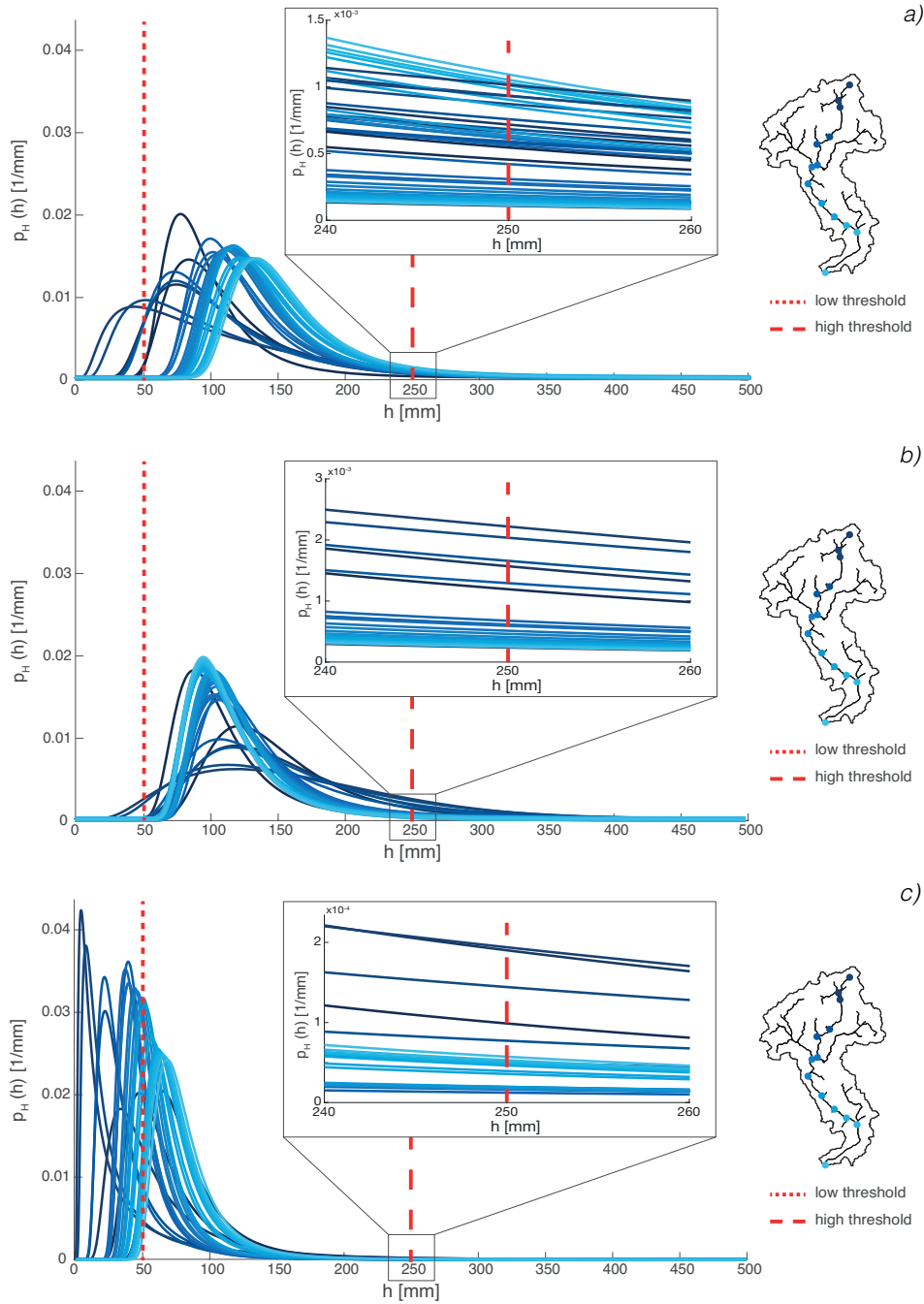
**Figure 3.5:** Stage pdfs along the network with  $\langle P \rangle = 150 \text{ mm/season}$  and a)  $\lambda_P = 0.1 \text{ d}^{-1}$ , b)  $\lambda_P = 0.5 \text{ d}^{-1}$  and c)  $\lambda_P = 1 \text{ d}^{-1}$ .

### 3.2. SPATIAL VARIABILITY OF HYDROLOGICAL VARIABLES

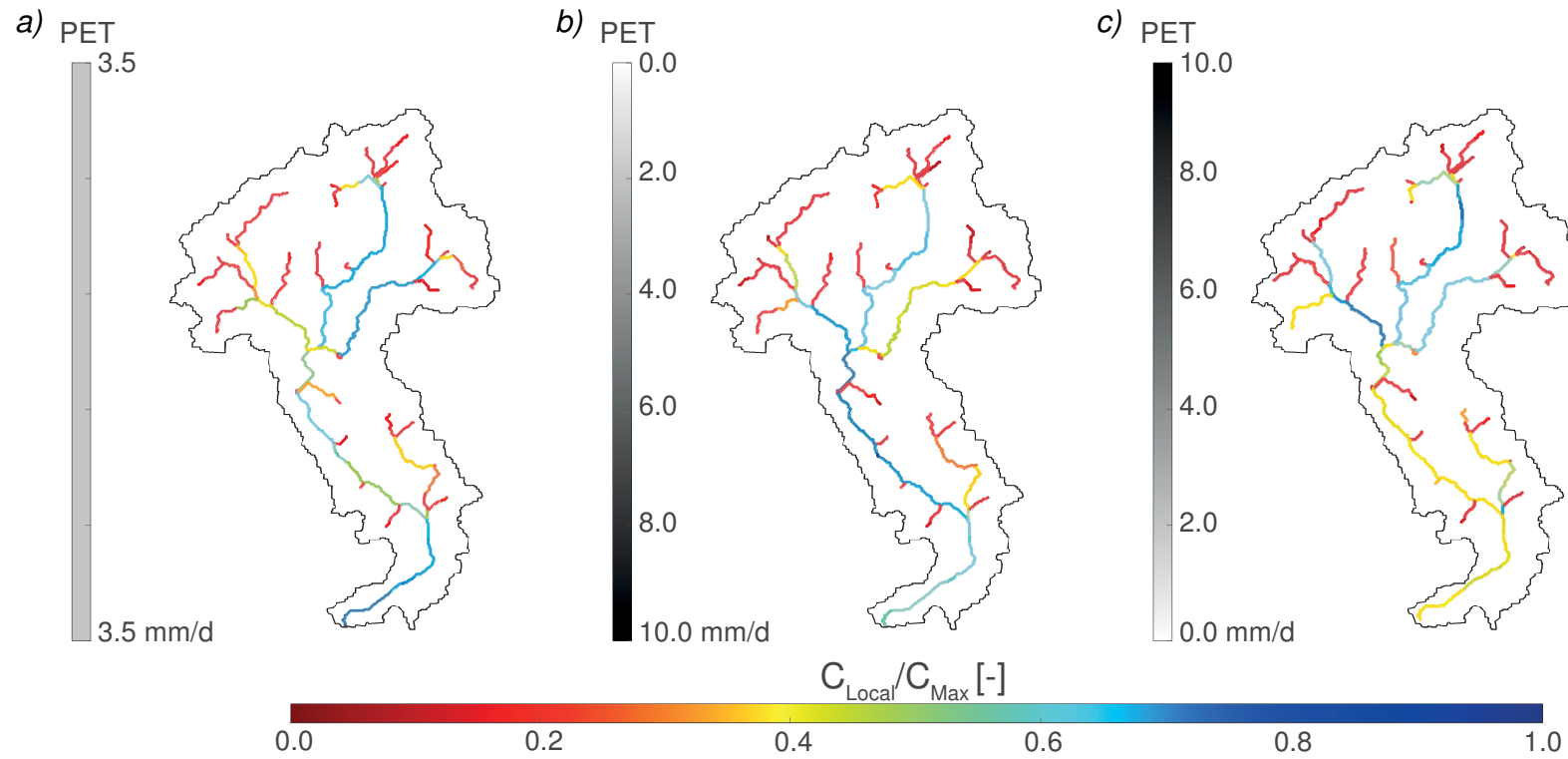


**Figure 3.6:** Spatial variability of local connectivity under a dry climate for increasing rainfall frequencies  $\lambda_P$  and assuming different stage thresholds  $h^*$ .

### 3. HYDROLOGICAL CONNECTIVITY



**Figure 3.7:** Stage pdfs along the network with  $\langle P \rangle = 150$  mm/season,  $\lambda_P = 1$  d $^{-1}$  and a) uniform  $\langle \text{PET} \rangle$ , b) North-South directed  $\langle \text{PET} \rangle$ , c) South-North directed  $\langle \text{PET} \rangle$ .



**Figure 3.8:** Normalized  $C_{\text{local}}$  assuming  $h^* = 250$  mm,  $\langle P \rangle = 150$  mm/season and a) uniform  $\langle \text{PET} \rangle$ , b) North-South directed  $\langle \text{PET} \rangle$ , c) South-North directed  $\langle \text{PET} \rangle$ .

# Chapter 4

## Habitat suitability

The spatial variability of streamflow regimes does not affect only river connectivity (equations (2.14) - (2.17)), but it also influences habitat distributions in rivers [Ceola et al., 2014; Jaeger et al., 2014; Santiago et al., 2017; Stamou et al., 2018]. This chapter describes the ecological function of river when habitat availability and the hydrologic alteration of flow regime are simultaneously accounted for.

### 4.1 Habitat suitability distribution

The ecological function of rivers relies on the presence of a mosaic of different habitats connected through the river network. However, the same ecological habitat can be usable for different ecological functions (or not) depending on the local streamflow availability. In this thesis, an empirical description of the ecological relevance of each node of the network is included. In particular, a local habitat suitability function that accounts for how the ecological functionality of a given site varies in time with streamflow is considered (Figure 4.1). Habitat suitability curves are a simple tool that describes species habitat preferences under different flow conditions, summarizing the effect of environmental variables on species distribution in rivers [Jowett and Richardson, 1990; Vismara et al., 2001]. In this work, an empirical gamma function is used to model the relation between fish habitats and flow availability [Fabris et al., 2017]:

#### 4.1. HABITAT SUITABILITY DISTRIBUTION

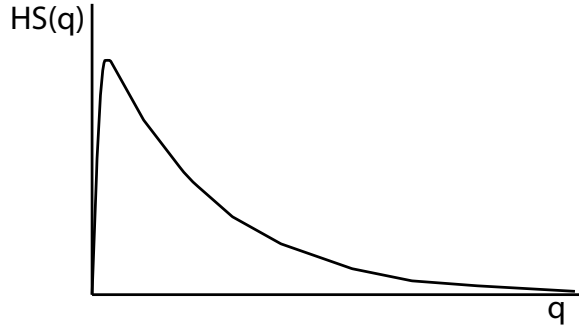
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$$HS(q) = C \exp(-B q) q^{A-1}, \quad (4.1)$$

where A, B and C are empirical parameters dependent on the channel morphology, water temperature and species length. The average value of the habitat suitability,  $\langle HS \rangle$ , is then obtained as:

$$\langle HS \rangle = \int_0^{\infty} HS(q) p_Q(q) dq. \quad (4.2)$$

Equation (4.2) quantifies the average ability of a given site to provide usable habitats under time-variant flow conditions, taking into account the local flow regime. Therefore, climatic and landscape variables affect, through  $p_Q(q)$ , both the connectivity along the network (equations (2.14) - (2.17)) and the average habitat suitability of each node (equation (4.2)).



**Figure 4.1:** Example of habitat suitability rating curve

In order to evaluate the ecological function of the hydrological connectivity the habitat distribution is also included in the simulation via equation (4.1), with specific reference to salmon migration towards the headwaters. Several studies pertaining Atlantic salmons, based on experimental data and hydraulic models [Tetzlaff et al., 2007; Fabris et al., 2017], have proposed a range of variability for the parameters A, B and C in equation (4.1). The minimum and the maximum



value for each parameter are reported in Table 4.1. In view of the uncertainty associated to the parameters controlling habitat suitability distribution, which depends on bed morphology, water quantity, water quality and size species, intermediate values of literature ranges are considered. Parameters are here assumed constant throughout the river network. Different parameters combinations were also tested without significant changes in the results presented in this thesis.

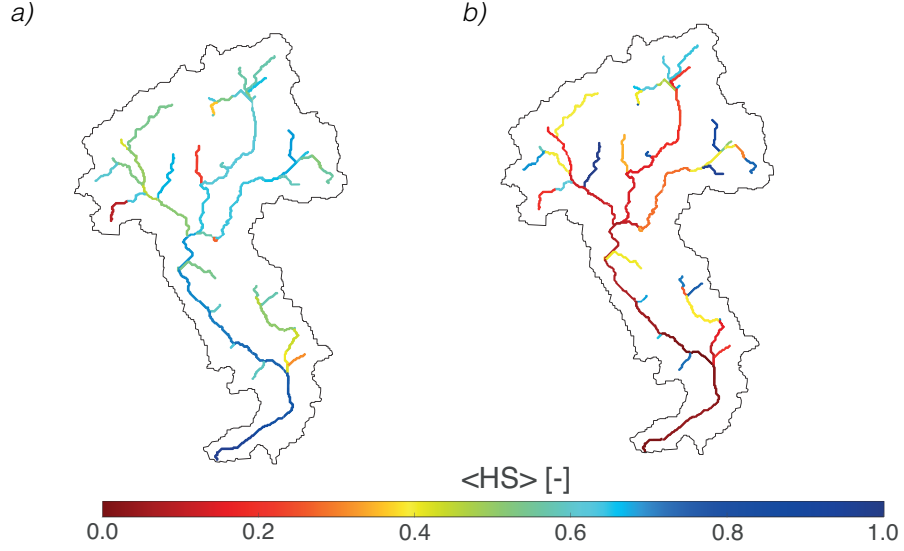
**Table 4.1:** Minimum and maximum values of habitat distribution model. The parameter set used in the simulation is also reported in the table (penultimate column)

Parameter	Min	Max	Assumed	Units
A	0.9	1.1	1.0	[-]
B	2.5	12.0	5.6	[L T <sup>-1</sup> ]
C	0.5	1.3	1.3	[T L <sup>-1</sup> ]

## 4.2 Ecological value of hydrological connectivity

This section investigates how the habitat availability and the hydrological connectivity interact in response to space-time variability of climatic attributes. As a proof-of-concept, it is presented here the specific example of spawning sites for Atlantic salmons, and their connectivity with the catchment outlet. The spatial distribution of spawning sites for Atlantic salmons is modelled according to the assumptions discussed in the previous section. Under these assumptions, the mean habitat suitability  $\langle HS \rangle$  is strongly dependent on mean precipitation and stream-flow. For very arid climates ( $\langle P \rangle = 150$  mm/season) downstream reaches are more suitable for spawning, whereas under wetter climatic conditions the higher habitat suitability is located in the headwaters (Figure 4.2).

To investigate the interaction between the spatial distribution of fish habitat suitability and hydrological connectivity as driven by flow regimes, the concept of outlet connectivity, which is a useful metric to evaluate the accessibility of spawn-



**Figure 4.2:** Spatial variability of mean habitat suitability under a) dry climatic scenario with  $\langle P \rangle = 150$  mm/season; b) a wet climatic scenario with  $\langle P \rangle = 350$  mm/season.

ing sites from the outlet in a river network is introduced. The outlet connectivity,  $C_{out}$ , is calculated using equation (2.16) with specific reference to the outlet.  $C_{out}$  expresses the probability for the outlet to be connected with all network nodes, assuming that each node represents a patch of suitable habitat where individuals can reproduce and survive. Thus, direct connections between nodes are migration links between patches. However, in order to preserve the ecological function of rivers, the hydrological connectivity must be guaranteed especially in those nodes whose ecological value is larger (in the example, the nodes where the habitat suitability is larger). Thus, the ecological connectivity of the outlet,  $C_{eco}$ , can be calculated by weighting each outlet-node path using a weight proportional to the mean habitat suitability of the node:

$$C_{eco} = \frac{\sum_{\substack{n=1 \\ n \neq out}}^N C_{(n \rightarrow out)} \langle HS \rangle_n}{\sum_{\substack{n=1 \\ n \neq out}}^N \langle HS \rangle_n} \quad (4.3)$$

Then, a suitability index  $SI$  is used to assess the impact of hydrological dynamics on ecological processes.  $SI$  is calculated as the ratio  $C_{eco}/C_{out}$ . If most ecologically

suitable sites are located in nodes that are mostly connected with the outlet,  $SI$  is larger than 1. On the other hand,  $SI < 1$  when the most suitable sites are less hydrologically connected with the outlet. Key results of the application of the model under different climatic settings and connectivity thresholds are summarized in Table 4.2.

**Table 4.2:** Outlet connectivity, ecological connectivity and suitability index for dry and wet climatic setting as function of stage water thresholds.

Threshold	Climate	$C_{out}$	$C_{eco}$	$SI = C_{eco}/C_{out}$
[ <i>mm</i> ]		[–]	[–]	[–]
$h^* = 50$	Dry	0.80	0.80	1.00
	Wet	0.85	0.70	0.80
$h^* = 250$	Dry	0.05	0.07	1.40
	Wet	0.50	0.20	0.40

*Dry:*  $\langle P \rangle = 150 \text{ mm/season}$ ;  $\lambda_P = 0.5 \text{ d}^{-1}$

*Wet:*  $\langle P \rangle = 350 \text{ mm/season}$ ;  $\lambda_P = 0.5 \text{ d}^{-1}$

Low stage thresholds produce high values of connectivity at local scale, both in dry and wet climates, as the probability to observe water depths larger than 50 mm is relatively high everywhere in the network. Thus, the probability of the outlet to be connected with the other nodes is high ( $C_{out} \geq 0.80$ ). In this case, most suitable sites (that are located downstream when climate is dry or upstream when climate is wet) are properly connected to the outlet, and  $SI$  values are close to 1 under both climatic scenarios. With larger stage thresholds ( $h^* = 250 \text{ mm}$ ), the most connected stream reaches are generally located close to the outlet, where the mean water stage is higher. In this case, the outlet connectivity is severely reduced and  $C_{out}$  under the dry climate is ten times smaller than the value obtained under the wet scenario. Although the outlet is insufficiently connected with the entire network, when precipitation is low (dry scenario), the most suitable sites are effectively connected to the outlet since they are located downstream ( $SI > 1$ ). On the other hand, when precipitation is high, the most suitable reaches are located in the headwaters, which are poorly connected to the outlet. Thus,

## 4.2. ECOLOGICAL VALUE OF HYDROLOGICAL CONNECTIVITY

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even though the overall values of  $C_{eco}$  and  $C_{out}$  are higher than those obtained in the dry scenario, SI is smaller than 1. This implies that, for relatively high thresholds, spawning sites are less accessible under wet climatic conditions. This simple example shows that, depending on the type of climate, the stage threshold and the spatial distribution of habitats, the emerging patterns of connectivity can either promote or limit ecological function of river networks.

# Chapter 5

## Ecological flows

This chapter illustrates the innovative water policy approach recently introduced in the European Union to manage water resource and its ecological function. The aim is to use the results of this thesis to propose a new perspective on the selection of a suitable methodology for ecological flows estimation in river systems.

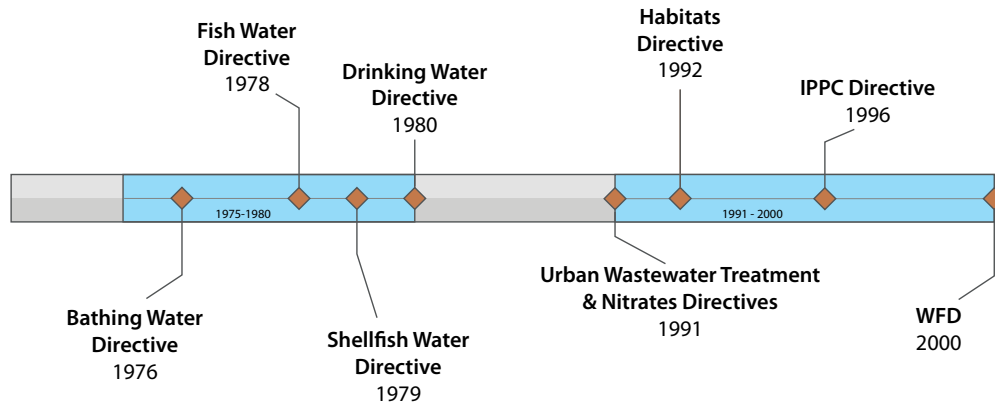
### 5.1 EU Water Framework Directive

Water legislation is one of the oldest and most developed areas of environmental policy in the European Union (EU) [Josefsson, 2012]. In the period between 1975 and 1980 (Figure 5.1), EU water policy focussed primarily on public health by setting Water Quality Standards (WQS) for the protection of water resources. They included Drinking Water Directive (Council Directive 80/778/EEC), a specific tool for drinking water abstractions from surface waters, as well as Directives for bathing waters (Council Directive 76/160/EEC), fish waters (Council Directive 78/659/EEC) and shellfish waters (Council Directive 79/923/EEC). A new phase of water legislation begun in 1991 with the adoption of the Urban Waste Water Treatment Directive (Council Directive 91/271/EEC) and the Nitrates Directive (Council Directive 91/676/EEC), restricting releases of pollutant into the aquatic environment. Then, a Directive for the conservation of natural habitats was adopted in 1992, addressing the maintenance of biodiversity, followed by a

## 5.1. EU WATER FRAMEWORK DIRECTIVE

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Directive for Integrated Pollution and Prevention Control (IPPC) adopted in 1996 and related to pollution from large industrial installations.



**Figure 5.1:** EU water policy evolution towards the WFD (1975-2000).

Although considerable progress had been made in tackling such individual issues, the need to overcome fragmentary water policy and establish a single piece of framework legislation emerged [Giakoumis and Voulvoulis, 2018]. In response to this, the European Union adopted the Water Framework Directive (WFD) in December 2000, with the following key goals:

- expanding water protection goals to all EU water bodies, both surface waters and groundwaters;
- achieving good status for all waters by a set deadline;
- adopting a river basin approach;
- developing an integrated water management system;
- granting public participation in water management;
- getting the prices right.

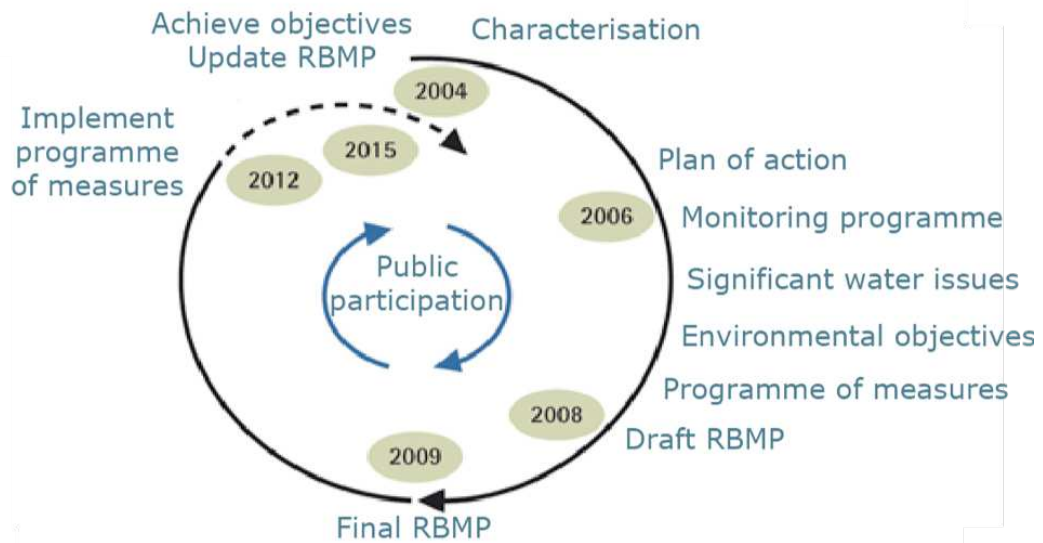
The WFD introduces a new framework for the management, protection and improvement of the quality of water resource and ensure its long-term and sustainable use across the European Union. The directive prescribes to Member States to achieve good status in all surface waters and groundwaters by 2015. In the case of grounds for derogation achievement of good status may be extended to

2021 or by 2027 at the latest. The directive establishes an innovative approach for water management based on river basins and sets specific deadlines for Member States to protect aquatic ecosystems. The key milestones of the water directive are listed below (Table 5.1). The implementation of the directive is achieved through

**Table 5.1:** Key milestones of Water Framework Directive

Year	Issue	Reference
2000	Directive entered into force	Art. 25
2003	Identification of River Basin Districts and Authorities	Art. 3
2004	Characterization of river basin: pressures and impacts	Art. 5
2006	Establishment of monitoring network	Art. 8
2008	Present draft river basin management plan	Art. 13
2009	Finalize RBMP including programme of measures	Art. 11, 13
2010	Introduce pricing policies	Art. 9
2012	Make operational programme of measures	Art. 11
2015	First management cycle ends	Art. 4
2021	Second management cycle ends	Art. 4, 13
2027	Third management cycle ends, final deadline for meeting objectives	Art. 4, 13

river basin planning processes that involve public participation in a series of steps shown in Figure 5.2. The river basin planning process started in 2004 with the analyses of the pressures and impacts affecting waters in the river basin district. The findings were used to define the environmental objectives and establish monitoring programs for improving water environment. The reports on Significant Water Management Issues (SWMIs) in 2007 were important steps leading towards the production of the first River Basin Management Plan (RBMP) in 2008. The directive requires via the RBMPs a Programme of Measures (PoM) to improve the ecological quality of water bodies and achieve environmental objectives. The PoM was published in December 2009 and was implemented by December 2012.



**Figure 5.2:** The WFD river basin planning process [ETC/ICM, 2012].

One of the main goal of EU water policy is to ensure sufficient quantity of good quality water for the environment and for people needs throughout Europe. According to the WFD, a good status is reached when certain standards have been met for the ecology, chemistry, morphology and quantity of waters [ETC/ICM, 2012]. The directive provides the definition of the status as the state of the system with minimal anthropogenic pressures or biological deviation from undisturbed conditions [EC, 2015]. The directive defines "good ecological status" and "good chemical status" in terms of healthy ecosystems as well as low levels of chemical pollution. The ecological status is related to the quality of biological communities, supported by hydrological and morphological characteristics of water bodies. Because of the ecological variability of different water types, which may be characterized by distinct definitions of environmental metrics, good ecological status cannot be defined across Europe using absolute standards. On the other hand, the chemical status is based on the compliance with all the quality standards established for chemical substances at European level.

In the last decades, climate change is posing a major challenge for water manage-



ment across the European Union [EC, 2008]. In southern Europe, low rainfall and high summer temperature are putting stress on environments with limited water resources, increasing the risks of eutrophication in many rivers, lakes, and coastal waters. On the contrary, more rain and a higher flood risk are observed especially in northern countries. As a result, the ecological and chemical status of EU waters is threatening, and the water ecosystems may become more vulnerable to extreme events. Therefore, policies and actions need to be set up in order to prevent and mitigate water scarcity and extreme situations, with the priority to move towards a water-efficient and water-saving economy of good quality waters.

Rivers flow on through different countries to reach the sea and they do not obey to national boundaries. A river basin (i.e. catchment) surrounds the entire river system, from the sources to the mouth. Therefore, water quality and quantity improvements cannot be reached at the local level without taking into account what happens upstream and downstream. For this purpose, the water directive proposes an integrated river basin management approach to protecting the whole body of water.

The river basin is the basic, natural, geographical and hydrological unit to manage water. Europe has more than 127 000 surface water bodies: 80% of them create an extensive network of rivers and streams, 15% are lakes and 5% are coastal and transitional waters [ETC/ICM, 2012]. The WFD divides the river basins and associated coastal areas into 110 river basin districts, 40 of which are international and cross borders, covering about 60% of European continent (Figure 5.3).

The WFD implementation in the EU Member States takes place through river basin management plans (RBMPs). The RBMPs identify measures to improve water quality and achieve the objectives set for the river basin (ecological status, quantitative status, chemical status and protected area objectives) within the timetable required. For each river basin district, the plans define the bodies of water not meeting the directive's environmental targets and describe causing problems and risks. They contain also the actions to be carried out to maintain and

## 5.1. EU WATER FRAMEWORK DIRECTIVE



Figure 5.3: Map of river basin districts and sea regions [ETC/ICM, 2012].

improve the quality of the water. Member States are called to measure the health of their surface waters and groundwater using national monitoring programmes. The monitoring is designed to provide a coherent and comprehensive overview of the health of European waters and shall permit a common classification of water bodies. The directive sets a five-class scale (high, good, moderate, poor and bad status) for surface waters and 2 classes (good and poor) for groundwater. The monitoring network helps Member States to define the effective measures needed to restore water bodies and achieve good status within each river basin. It is worth to note that the directive sets a common approach for monitoring water quality across all Member States but it does not specify the methods to be used. Indeed, every Member State decides the best method based on local conditions and existing national approaches. The directive specifies three types of monitoring [EC, 2008]:

- Long-term surveillance monitoring, which provides a broad understanding of the health of water bodies and assess long-term changes in natural conditions and resulting from widespread anthropogenic activities;
- Operational monitoring, which focuses on water bodies identified as being at risk of failing to meet their environmental objectives;
- Investigative monitoring, which aims to ascertain the magnitude and impacts of accidental causes.

Nowadays, aquatic ecosystems health is a primary objective for European water policy as previous legislation were mainly focused on chemical pollution. While Member States have a great deal of experience in monitoring the chemical status of their waters, measuring good ecological status is a new challenge. The WFD requires that the national classification systems for assessing ecological status should be intercalibrated. The aim of the intercalibration is to ensure that the good status class given by each national assessment methods is comparable and consistent with the directive. Given the wide range of ecosystems found across European territory, using one single method to assess all water bodies is limiting. Therefore, intercalibration provides a common scale across Europe to measure progress towards healthy ecosystems. To date, eleven countries participate in the intercali-

## 5.1. EU WATER FRAMEWORK DIRECTIVE

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bration group for North-East Atlantic coastal and transitional waters, comparing the ecosystems of seven different types of waters, from shallow coastlines to deep northern fjords [EC, 2008].

The new water directive encourages active involvement of all interested parties using public information and consultation in the process of water management. The public participation is essential to gain knowledge and experience from stakeholders and jointly develop solutions to environmental problems. This consultation mechanism leads to shared decision-making in which stakeholders actively participate in the development and implementation of river basin plans.

The directive introduces key economic principles for the management of European waters. Water services, such as supplying clean drinking water, irrigation for agriculture, hydropower production and wastewater treatment facilities, must be charged for the services provided. The prices paid by users should also cover resource and environmental costs. For instance, environmental costs may include damage to ecosystems by extracting water for anthropogenic uses and reducing water levels in rivers. Recovering resource costs is especially important in river basins where water is limited. Nowadays, water scarcity and droughts are a growing concern throughout Europe [EC, 2008]. Water scarcity, where demand exceeds the sustainable use, affects over 10% of the EU's population and almost 20% of its territory. Moreover, the number of droughts due to low rainfall has increased over the past 30 years and in 2003 they affected over 100 million people across Europe. The directive also states that water pricing should create incentives for the efficient and sustainable use of water resources. The idea is that, if users pay the real costs of used water they would certainly reduce water losses. Pricing would be a powerful awareness-raising tool for consumers, as it combines environmental with economic benefits.

The Water Blueprint highlights that preserving water is not only about environmental protection, health and well-being, but it is also about economic growth and prosperity. The Blueprint outlines actions for the implementation of current

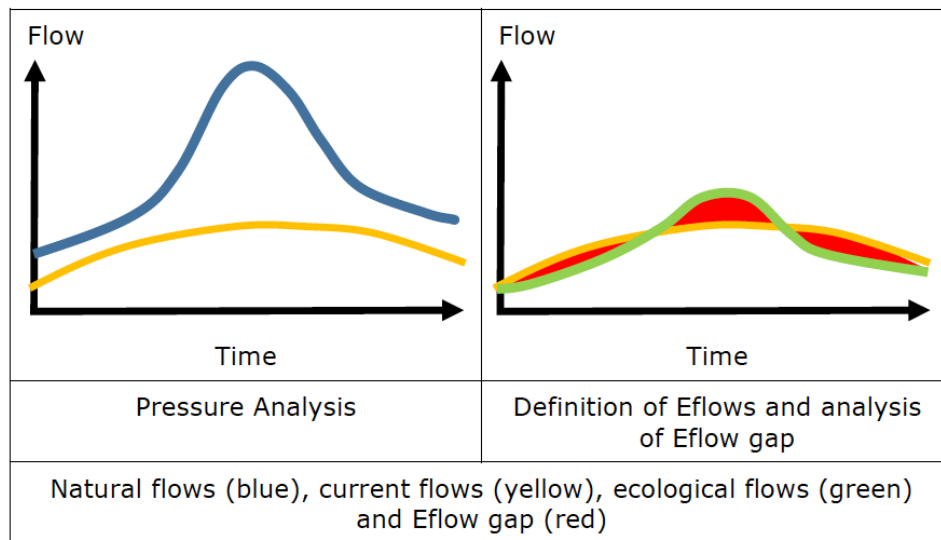
water legislation integrating policy objectives related to water quantity and efficiency. A sufficient quantity of good quality water must be available for people's needs, the economy and the environment throughout the EU. In response of this, water allocation needs to be improved on the basis of the amount of water required for the aquatic ecosystem to continue to thrive and provide the services we rely upon [EC, 2015]. Fundamental to this is the recognition that water quality and quantity are intimately related within the concept of good status demanded from the directive. To achieve this, the Blueprint proposed the development of a guidance document in the framework of the water directive (Common Implementation Strategy, CIS) that would provide an EU definition of ecological flows and a common understanding of how it should be calculated.

## 5.2 From environmental flows to ecological flows

The concept of environmental flows was historically developed as a response to the degradation of aquatic ecosystems caused by the overexploitation of water. Terms such as minimum flows, instream flows and fish flows were introduced to identify the requirement of a minimum amount of water that should continue flowing along rivers to sustain biodiversity and ecosystem integrity, even in the dry periods [Cavendish and Duncan, 1986; Milhous et al., 1989]. Environmental flow is defined by the Brisbane Declaration [2007] as “the quantity, quality and timing of water flows required to sustain freshwater ecosystems and the human livelihoods and well-being that depend on these ecosystems”. The directive acknowledges the critical role of water quantity and dynamics in supporting the quality of aquatic ecosystems. It considers the ecological flow (Eflow) as “a hydrological regime consistent with the achievement of the environmental objectives of the WFD in natural surface water bodies”. The hydrological regime plays a primary role in determining physical habitats, which in turn determines the biotic composition and sustainability of aquatic ecosystems [Junk et al., 1989; Poff et al., 1997; Bunn and Arthington, 2002; Arthington et al., 2006; Poff and Zimmerman, 2010]. The assessment of the hydrological regime is explicitly required by the di-

## 5.2. FROM ENVIRONMENTAL FLOWS TO ECOLOGICAL FLOWS

rective for the classification of ecological status related to all existing pressures, in particular to hydrological ones. In water bodies affected by significant hydrological pressures (i.e. land-use and drainage modification, water extraction and impoundments), the gap between the current flow regime and the ecological flow should be assessed in order to set appropriate mitigation measures. Figure 5.4 shows in detail the difference between the pressure analysis and the gap analysis. While the pressure analysis (left) assess the hydrological alteration considering the deviation of current flows from natural flows, Eflow gap analysis consists in assessing the distance between current flows and ecological flows. This gap analysis requires an estimation of the ecological flow taking into consideration the natural flow regime, the morphology of the river and the ecosystems therein.



**Figure 5.4:** Pressure analysis and Eflow gap analysis [EC, 2015]

Several methods have been developed in the scientific field to inform the definition of Eflows, mostly differing in terms of integration of biological aspects, scale, complexity and volume of data requirement. These methodologies can be grouped in three general categories: (1) Hydrological, (2) Hydraulic-Habitat, and (3) Holistic methodologies [Tharme, 2003]. The following sections describe such methodolo-

gies; the selection of the most appropriate method depends on resource availability and on the severity of human pressures.

### **Hydrological methodologies**

Hydrological methodologies rely primarily on the use of hydrological data, usually historical monthly or daily flow records, for regional assessments. From the 1970's purely hydrological methods set minimum flow level to be maintained to protect the aquatic resources [Tennant, 1976]. More recent applications are moving towards more comprehensive methodologies that recognize the importance of natural flow variability and suggest bands of allowable alterations from natural conditions, e.g. the Sustainability Boundary Approach (SBA, [Richter et al., 2012]). These approaches allow to find flow levels that naturally occur in rivers and can be considered allowable thresholds without compromising ecological health and ecosystem services. With reference to Figure 5.4, they assume Eflows (green line in right panel) overlaps to natural flows (blue line in left panel).

Hydrologically-based methods currently represent the most widely used approaches for Eflows estimation because of their ease of use and low cost, since fieldwork is not needed. However, these approaches do not directly include any ecological and morphological characteristics of rivers nor any biological processes that occur therein.

### **Hydraulic-Habitat methodologies**

Because of the lack of sensitivity to individual rivers pertaining to hydrological methods, hydraulic-rating methods are subsequently developed to describe channel-discharge relationships by using field measurements. Though providing river-specific data, these methodologies failed to explain the implication of changes in the physical conditions for the aquatic biota. This led to the development of habitat-rating approaches, the best known of which are the Instream Flow Incremental Methodology IFIM [Bovee, 1982] and its cornerstone, the physical habitat simulation model, PHABSIM [Milhous et al., 1989]. These models are grounded on the idea that changes in fish habitat are due to changes in flow conditions.

Hydraulic-habitat simulation consists of i) physical modelling of the river channel and ii) modelling of the biological associations with the physical environment. The physical and biological models are combined to simulate how the variability of physical habitats (e.g., the wetted area suitable for a target species) varies with streamflow. For this reason, the habitat time series analysis is currently considered a key component in the definition of Eflows with hydraulic-habitat methodologies [Parasiewicz et al., 2013].

These approaches represent an earlier new scientific front to understand the ecological consequences of environmental flows on habitats and biota, but it is worth to note that the hydraulic-habitat simulation methods estimate only the amount of habitat as a function of hydrological and morphological conditions at local scale. Moreover, hydraulic-habitat methodologies require a significant amount of fieldwork to collect both the hydro-morphological and biological data; thus, they can be time consuming and expensive.

### **Holistic methodologies**

Holistic methodologies aim to prescribe flows for the maintenance of the whole riverine ecosystem [Zalucki and Arthington, 1998]. Holistic approaches are processes that allow multi-disciplinary scientists (with various backgrounds including hydrology, geomorphology, water quality and ecology) to integrate data and knowledge for developing an understanding of the relationship between flow alteration and environmental response. The output is a description of the altered flow regime needed to achieve and maintain components of the river ecosystem, including societal and recreational uses. Indeed, these frameworks may also integrate social, cultural and economic values within environmental protection goals (e.g. BBM [Tharme and King, 1998]; DRIFT [King et al., 2003]; ELOHA [Poff et al., 2009]). Such methodologies address the flow requirements of the entire riverine ecosystem, based on explicit links between changes in flow regime and foreseen effects on the biophysical environment connected to the river network.

Holistic methodologies are generally applicable for regional or river specific scales. Depending on the depth of the evaluation, data collection, and the extent of ex-



pert consultation, their applications can be time consuming and very expensive. Though the principles on which they rely are robust, these workshops provide little quantitative information and they are difficult to translate into the engineering practice. Therefore, they are seldom applied for the design of water infrastructures and the identification of operational management criteria.

### 5.3 A new perspective on the Eflows evaluation

Understanding whether deviations from the natural flow regime are acceptable or not is a challenging problem. This thesis aims to stress out the importance of an holistic perspective that integrates hydrological, hydraulic and physical habitat analysis, within a modular framework, for establishing the environmental requirements of riverine ecosystems. In particular, this work proposes an innovative approach which is based on the probabilistic method for the identification of spatial patterns of ecologic variables integrated in the long-term at network scale.

It is proposed that, in order to preserve flow regimes for supporting the aquatic ecosystems, it would be convenient to take into account the network-scale function of the natural hydrologic regime. The spatially-integrated approach proposed in this thesis considers individual local flow conditions, the associated habitat, and their interconnections, thereby overlapping the multiple functions of river systems. Hydraulic conditions are relevant as they guarantee habitat integrity at local scale. However, the key point is to ensure valuable habitats which are physically reachable and hydrologically connected. This can be feasible considering physical thresholds, which are embedded in the concept of the critical stage  $h^*$ , defined as the minimum stage that allows species movement from one point to another within the river network.

The results of this thesis demonstrate that temporal and spatial variability of flows reflects in a variability of the hydrological connectivity, and thus might imply seasons with a strongly reduced connectivity. These represent a barrier for the movement of ecological species within the catchment, with associated environmental concerns. Migrations are biologically constrained in time and physical dis-

### 5.3. A NEW PERSPECTIVE ON THE EFLOWS EVALUATION

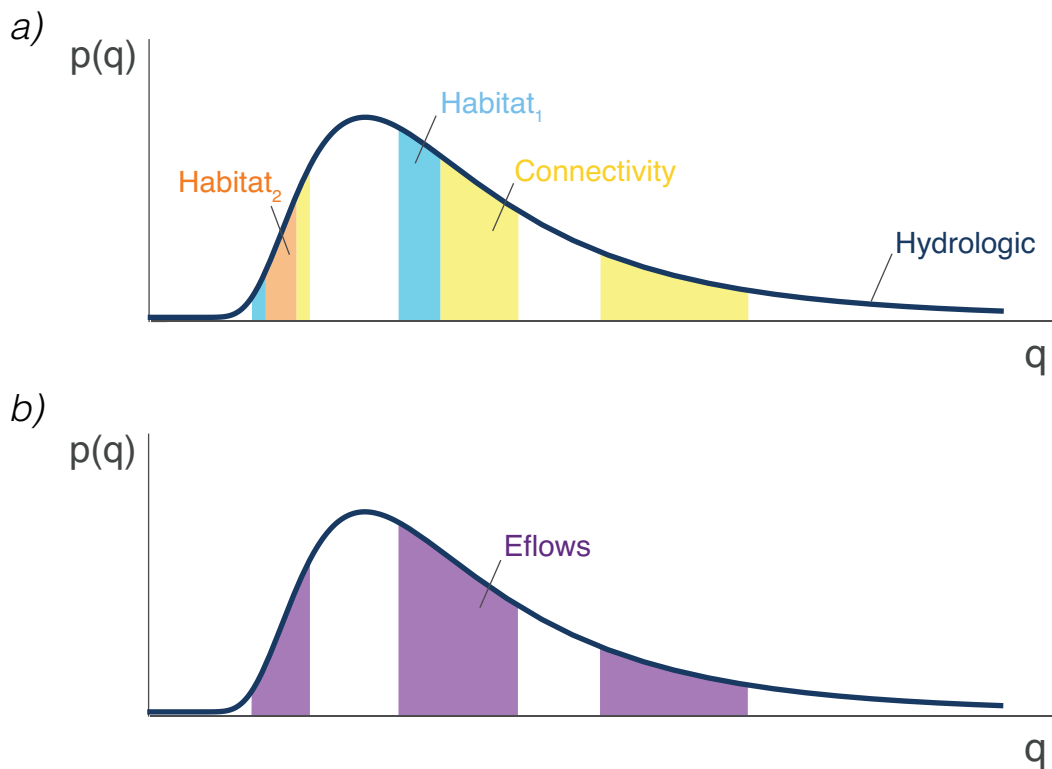
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connections between habitats during prescribed time windows might be extremely harmful for fish communities. In the case of salmon upstream migration, recent studies have demonstrated that the lack of hydrological connectivity can reduce the number of immigrating salmon by up to 80 % of the potential value under optimal hydrologic condition [Lazzaro et al., 2017].

Hydrologic variability controls both the existence of unique and relevant habitats and the connectivity between these habitats along the river network. Thus, these two aspects cannot be decoupled in the protection of aquatic communities. Moreover, local drops of connectivity induced by anthropogenic pressures, such as damming or water diversions contribute to the fragmentation of the overall river network [Widder et al., 2014], disconnecting different type of habitats supplied by the river system. Frequently, dams and diversions cause dramatic changes in downstream flow regimes and might limit the access to fundamental habitats hosted by upstream reaches. The impact of water infrastructures, superimposed to flow regimes alterations induced by climate and landscape change [Botter, 2014], can make considerable portions of the network unavailable to fishes and biomes due to connectivity losses in critical nodes of the river network.

Many regions of the world are experiencing an increasing exploitation of riverine water resources. Therefore, preserving the hydrological connectivity within a river network and, consequently, preventing environmental concerns on biological communities are becoming increasingly important tasks for water managers. Following the recent water directive provisions, anthropogenic and environmental needs have to be balanced, thereby requiring tools for a proper allocation of water resources among contrasting goals. The comprehensive view of the river system proposed with this thesis provides a clue for the identification of river flows that guarantee the functioning of all major biotic components of ecosystems. Pure hydrological methods seek to preserve the frequencies associated to all flow magnitude, according to the natural flow regime paradigm. Habitat methods suggest classes of “critical” environmental flows consistent with the ecological requirements of rivers, which may be dependent on the species considered. As the habitat conservation is a necessary but not sufficient condition to sustain river ecosystem function, in the

definition of ecological flows, also river flows that ensure a sufficient connectivity need to be accounted for. Figure 5.5 shows that habitat requirements in terms of hydrological conditions may differ from connectivity requirements (panel a).



**Figure 5.5:** a) Distribution of natural flows, with highlighted the frequency associated to flows that ensure ecological habitats for different species (orange and cyan) and network connectivity (yellow); b) Distribution of natural flows, with highlighted the frequency associated to flows that ensure the ecological flow.

The final choice to define ecological flow requirements is a multifaceted process that involves prioritization of water uses, knowledge of the replenishing water resources and risk assessment efforts. Key decision variables are the presence of habitats and the connectivity. These goals may compete when suitable habitats

### 5.3. A NEW PERSPECTIVE ON THE EFLOWS EVALUATION

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are not physically connected under certain flow conditions. In this context, multi-objective optimization may be useful tools to address the evaluation of Eflows and identify optimal trade-offs between the maximization of the ecological function of water resources and the internal connectivity of networks.

# Chapter 6

## Discussion

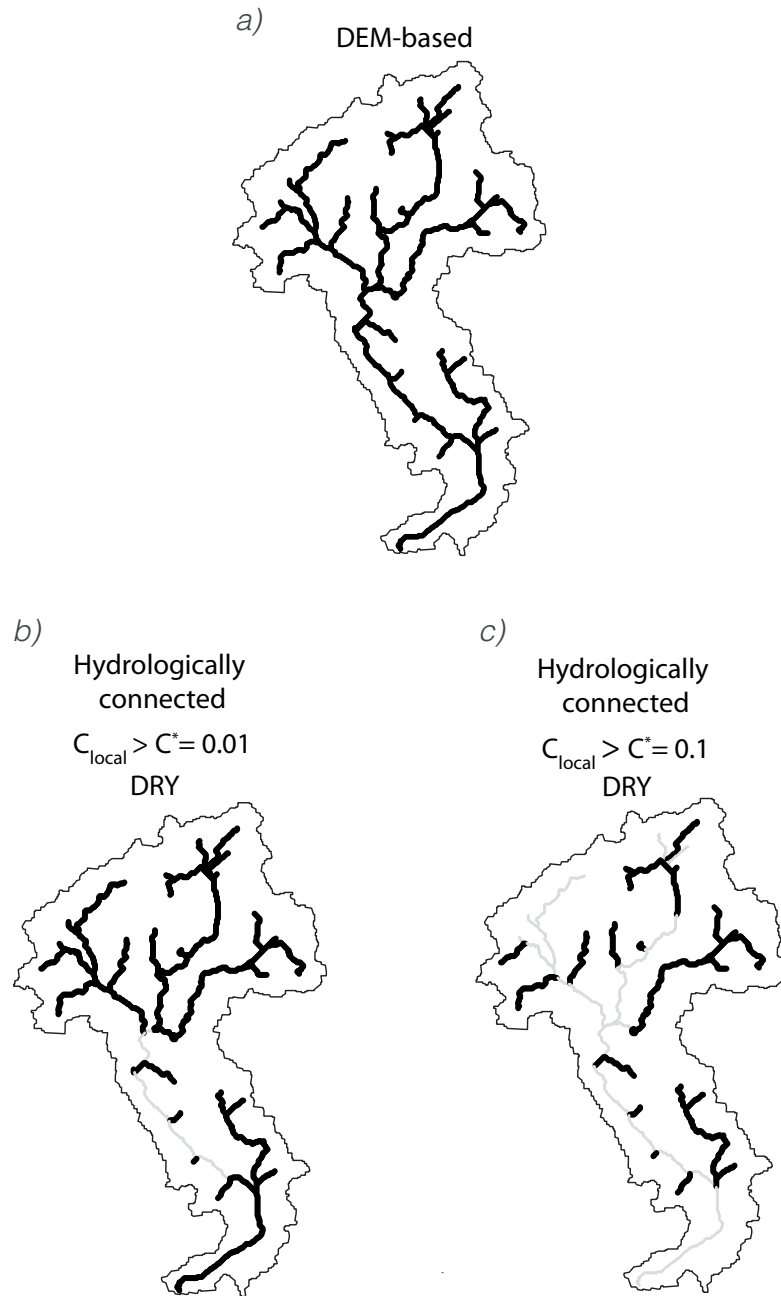
The ecological function of rivers is guaranteed by the physical connection between network nodes, which is driven by hydrological processes. The results of this thesis indicate that the spatial variability of reach-scale connectivity might be controlled by the spatial and temporal distribution of climatic variables. Precipitation distribution, in terms of rainfall frequency and intensity, and spatial patterns of evapotranspiration concur to define the fraction of the hydrological network available for biological dispersion. Frequently, river networks in arid environments may be hydrologically disconnected because of insufficient water flows in relevant portions of the network. Moreover, spatial gradients of climatic properties influence the hydrological response and the connectivity of catchments whose size is larger than the integral scale of the relevant climatic heterogeneity. Therefore, spatial patterns of climate are likely to alter existing scaling properties of drainage networks inferred through purely geomorphological approaches [Rigon et al., 1993].

Although the general influence of the hydrological connectivity on fauna migratory dynamics has been already documented in the literature [Tetzlaff et al., 2008; Jaeger et al., 2014; Lazzaro et al., 2017], quantitative assessments of ecologically relevant stage thresholds remain problematic. In this novel framework, a critical connectivity threshold can be introduced to identify the likelihood of hydrological conditions favourable to migratory movements. In particular, it shall assume that two nodes with a local connectivity lower than a given threshold,  $C^*$ , are physically

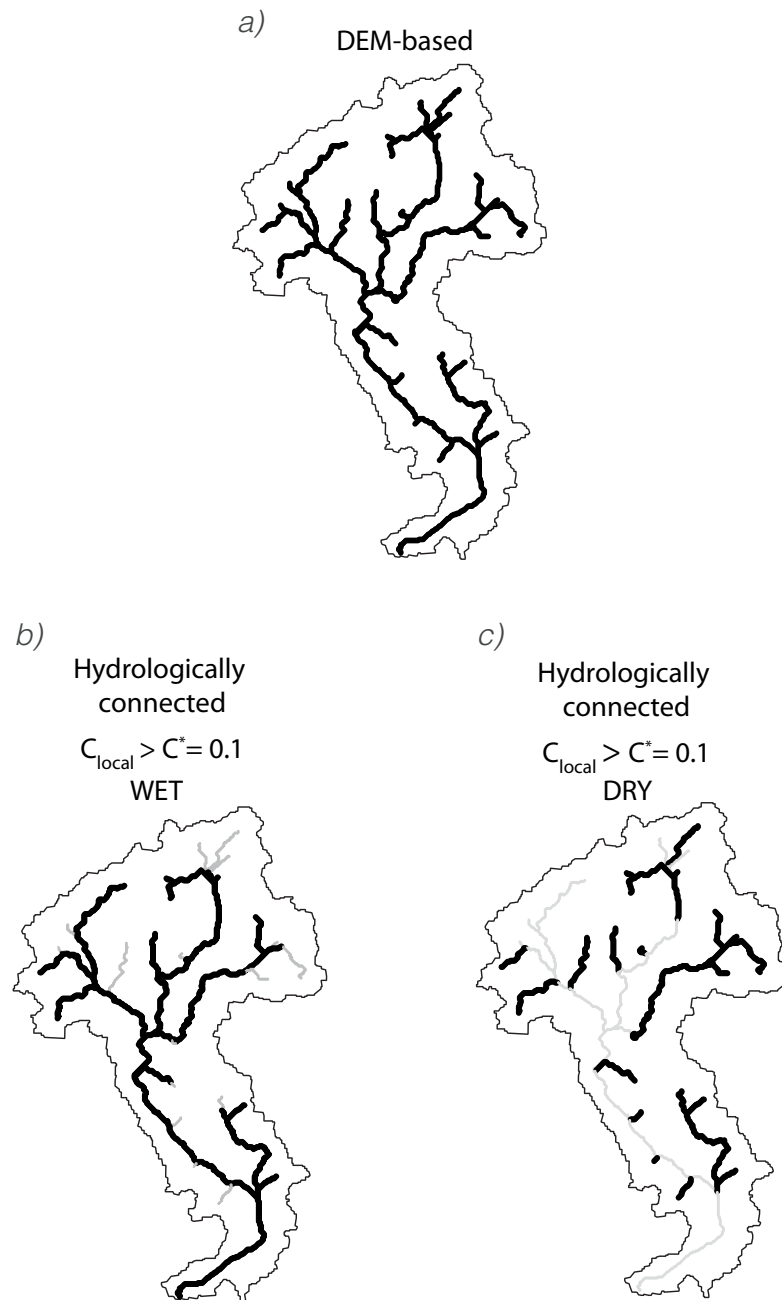
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disconnected because the likelihood of hydrological conditions favourable to species movement is too low. Lower values of  $C^*$  are thus associated to greater efficiency during migration. Simulations evidence that the shape of the connected network might be significantly altered by the underlying hydrological processes. This is represented in Figures 6.1 and 6.2, that show the shape of the connected network under different scenarios, whenever all the reaches with  $C_{local} < C^*$  are removed from the original network. During the dry season a low connectivity threshold ( $C^* = 10^{-2}$ ) breaks the network into two disconnected parts. Larger portions of the main river channel are progressively excluded by increasing the connectivity threshold (Figure 6.1b and c). The shape of the network is also modified when different climatic conditions are considered (Figure 6.2b and c). Interestingly, under a wet climate the headwaters are disconnected; conversely, in the dry scenario the main channel, where the connectivity is lower, gradually disappears. This dynamic behaviour of the flowing network might have a crucial impact on ecological models for species dispersion and propagation of waterborne diseases [Rinaldo et al., 2018]. The proposed approach provides a quantitative framework that allows the description of the main hydrologic causes and ecological consequences of hydrological dynamics experienced by river networks in response to climatic forcing. As such, the method could be integrated into network transport models currently utilized in spatial ecology, allowing for the use of time-variant and locally disconnected network domains, of the type shown in Figures 6.1 and 6.2.

This work exploits a probabilistic framework for the characterization of the spatial variability of streamflow regimes and water stage dynamics driven by external climatic forcing. The method incorporates a number of hydrological models of proven robustness and wide applicability [Porporato et al., 2004; Botter et al., 2009; Biswal and Marani, 2010; Doulatyari et al., 2015, 2017]. Nevertheless, the model relies on a number of simplifying assumptions. The hydrological model assumes a one-to-one relationship between catchment storage and discharge, which is here inferred solely from geomorphic data. Moreover, the developed framework does not take into account space-time variations in the relationship between river width and depth along the network. The constant nonlinear relation used to de-



**Figure 6.1:** Comparison between DEM-based network and hydrologically connected networks. a) DEM-based network; b) Hydrologically connected network using connectivity threshold  $C^* = 0.01$  in dry climatic conditions; c) Hydrologically connected network using connectivity threshold  $C^* = 0.1$  in dry climatic conditions.



**Figure 6.2:** Comparison between DEM-based network and hydrologically connected networks. a) DEM-based network; b) Hydrologically connected network using connectivity threshold  $C^* = 0.1$  in wet climatic conditions; c) Hydrologically connected network using connectivity threshold  $C^* = 0.1$  in dry climatic conditions.



river water stages from discharges is an assumption that could be relaxed only whenever in-situ measurements of the geometry of cross-sections along the river are available. River bed is also assumed impermeable and possible interactions between the stream and the surrounding environment are neglected. Nevertheless, the model is mathematically sound, has a reduced number of parameters with a direct physical meaning, and it is computationally inexpensive. Therefore, the approach represents an interesting prospect for eco-hydrological spatially-explicit studies.



# Chapter 7

## Conclusions

Natural flow regimes display pronounced variability at different spatial and temporal scales. The magnitude, frequency and duration of streamflows are controlled by climatic forces, such as rainfall and evapotranspiration. In turn, the changing quantity of water flowing in a river significantly influences the connectivity among river stream reaches. This thesis proposes an analytical approach where hydrological connectivity is explicitly linked to driving hydroclimatic variables and catchment properties through the emergent spatial patterns of streamflow regimes along river networks. The method is based on a stochastic generation of rainfall able to reproduce different climatic scenarios in terms of rainfall frequency, intensity and amount. Results confirm that precipitation regimes significantly impact the connectivity of river networks. Network connectivity typically increases by increasing the mean precipitation and the frequency of rainfall events. Under arid climatic conditions, network connectivity is higher for rare but intense events, of the type found in semi-arid regions.

Evapotranspiration is a key factor controlling the rate of decrease of the runoff coefficient along river networks, with noticeable effects on mean water stages and hydrological connectivity. A smooth decrease of the rainfall-runoff coefficient with the contributing area generates increasing mean stages for larger drainage areas; vice versa, when the reduction of the runoff coefficient with the contributing area is faster (e.g. when rainfall events are frequent and when evapotranspiration is spa-

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tially variable, especially under arid conditions), the mean stage increases much slower downstream, making the connectivity dependent on the interplay between flow variability and the stage threshold  $h^*$ .

Model simulations show that spatial patterns of evapotranspiration strongly influence the variability of the hydrological connectivity along the network, without impacting the mean network connectivity.

The proposed framework helps to identify the physical controls on hydrological connectivity and their effect on ecological processes along river networks, as documented by the proof-of-concept pertaining to salmon migration discussed in section 4.2. The analysis shows that depending on the climate and the spatial variability of habitat suitability, the resulting connectivity patterns can either promote or limit the ecological function of river networks.

To provide a quantitative assessment of the impact of hydrological processes on the shape and the extent of connected reaches, changes in the topological configuration of the river network, when all the streams with insufficient connectivity are removed, are analysed. The analysis reveals that under arid climates the main channel may become disconnected from tributaries, whereas under wet climates, river networks tend to shrink from headwaters. Therefore, this thesis provides quantitative evidence of the fact that the shape of connected networks can be significantly impacted by the underlying hydrological dynamics.

The network-scale approach developed in this thesis offer an objective basis to explicitly account for the climatic controls on river flow regimes, the hydrological connectivity and the ensuing ecological consequences. The general mathematical formulation proposed here encourages the application to different types of synthetic networks (e.g. OCNs) and to real world case studies. The method offers a robust basis to assess ecological impacts of streamflow variability in rivers, and it is thus suited to be coupled with spatially-explicit ecological network models. The approach can also help the definition of ecological flow requirements within sustainable water policies.

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