



## Deliverable 4.1. Conceptual model of the ecosystem services provision of DRNs and its application at the focal DRN level

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## General Introduction

River networks cover a small fraction of Earth's surface (0.8%) and contain around 0.01% of the world's water. However, they harbour disproportionate levels of biodiversity, supporting 10% of all described species. They contribute substantially to global biogeochemical cycles, releasing greenhouse gasses into the atmosphere and transporting carbon and nutrients from continents to oceans. River networks also supply clean water for humans, and other ecosystem services (ES) such as flooding regulation, nutrient retention, or food provisioning. Despite their remarkable importance, growing evidence suggests the number of streams and rivers drying due to climate change is increasing. In addition, rivers and streams that naturally do not flow all year round represent a large fraction of flowing waters. However, how drying of River Networks affects the provision of ES at local and regional scales is still in its infancy.

This report presents a first part which develops a conceptual model of ecosystem service provisioning in Drying River Networks (DRN) and a second part with a modelling exercise estimating the provision

of a selection of key ES in the 6 DRNs of the project. The selection of ES incorporates Water provisioning, Flood regulation, Erosion Regulation, Drought Mitigation, Thermal regulation and Carbon emissions. Moreover, the modelling exercise provides a benchmark to assess and quantify how catchment and river network ecosystem service provisioning might be affected by climate change.

More specifically, in the first part the conceptual model integrates ecological, hydrological and landscape components to capture the dynamic nature of ES provisioning across catchments and river networks. It considers factors such as hydrological regimes, sediment and organic matter transport, and biotic interactions that influence a wide variety of ecosystem functions. By incorporating these factors, the model enables a comprehensive evaluation of the direct and indirect contributions of specific landscape components to ES provision.

In the second part, a modelling exercise for each DRYvER case study is presented to demonstrate the practical application of the conceptual model for the above selection of 6 ESs. The modelling approach uses spatial data from each local DRN to assess the current provision of ESs, identify key drivers of change and key environmental factors. Moreover, the results illustrate the potential for restoration or conservation actions within each DRN. These results could inform participatory processes for the design of adaptation pathways through implementation of Nature Based Solutions for climate change resilience.

The titles of each part of this deliverable are:

Part 1. Conceptual model for ecosystem services provision in Drying River Networks

Part 2. Ecosystem services provision under drying conditions and its application at the focal DRN level

### Keywords

Drying River Networks, Ecosystem Services, Drought, Hydrological Regimes, Integrated Catchment Management, Benchmark Ecosystem Service Provision, Conservation, Restoration, Adaptation Pathways

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R=Document, report (excluding the periodic and final reports)  
DEM=Demonstrator, pilot, prototype, plan designs  
DEC=Websites, patents filing, press & media actions, videos, etc.  
OTHER=Software, technical diagram, etc.  
ORDP : Open Research Data Pilot

<sup>2</sup> Use one of the following codes:

PU=Public, fully open, e.g. web  
CO=Confidential, restricted under conditions set out in Model Grant Agreement  
CI=Classified, information as referred to in Commission Decision 2001/844/EC.

# Conceptual model for ecosystem services provision in Drying River Networks (Part 1)

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## Abstract

The first part of this deliverable is divided into three main blocks. The first one presents preliminary concepts related to the ecosystem services (ES) framework. This is intended to lay a solid foundation of principles and terminology that will be used throughout the document. The second part presents the main physical and ecological dynamics that structure the ES provision in river networks and their catchments. This section is divided in two sections, each of which refers to spatial and temporal dynamics, respectively. Finally, the third block focuses on outlining the methodological basis that will be used for modelling ES in DRyVER and presenting the conceptual model for the ES provision in river networks. In the first section, we set out the general conceptual and digital framework used for ES modelling. In the second section, we conceptualise and integrate each ES into the framework described in the previous section, describing the main variables and spatio-temporal dynamics that control the provision of each ES. In the third section, we present the conceptual model of ES provision in river networks and their catchments. This model responds to the challenges of considering not only temporal dynamics, but also the multiple interactions between catchment processes and the river network ones.

### Keywords

Ecosystem service models, Catchment and river network interaction, Integrated catchment Management, Ecosystem service trade-offs, Landscape components

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## General framework to ecosystem services characterization

The ecosystem service (ES) paradigm encompasses the direct and indirect benefits that people obtain from natural capital (Potschin & Haines-Young, 2011). Currently, this framework is used by a wide range of stakeholders from multiple disciplines, including scientists, economists, policy makers and land managers, which often leads to some ambiguities in the use of its early concepts and definitions (Lamarque et al., 2011). This should not only be seen as a factor that makes it difficult to compare different projects, policy contexts, time and space, but may also reduce the potential for applying the ES framework to specific issues (Fisher et al., 2009). The DRYVER project aims at characterizing and valuing a set of ES, so an appropriate conceptualization should be provided to reduce ambiguity and to reach a common ground of discussion. ES characterization imply understanding some questions such as what ES are, where and how they are provided, by whom they are used or how they could be enhanced or eroded. This requires a clear and precise understanding of the foundations that support the ES framework (Fisher et al., 2009; Syrbe & Walz, 2012).

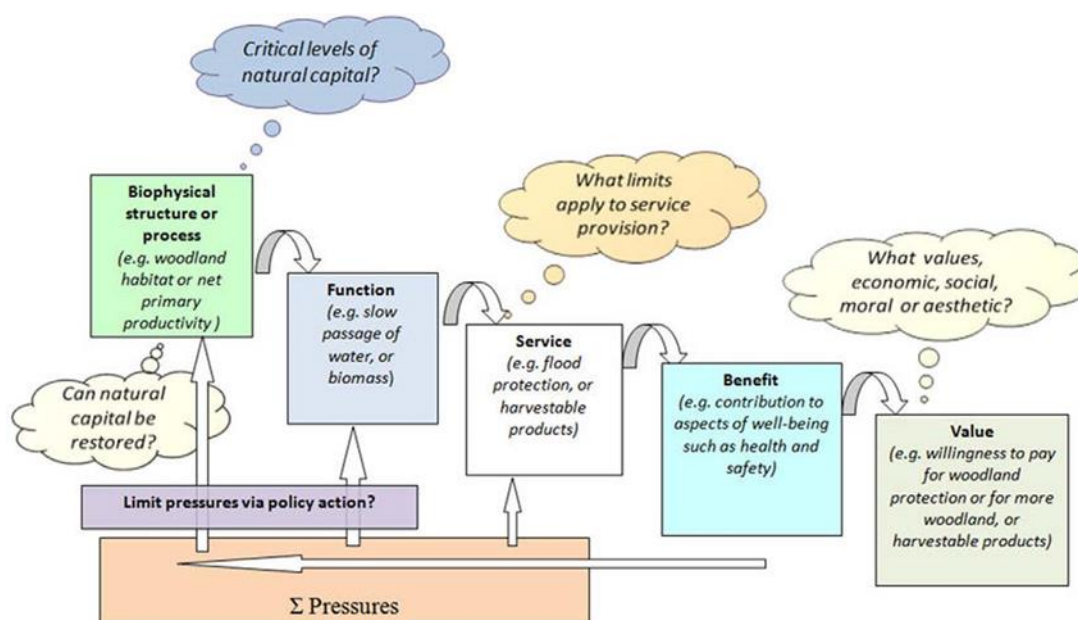
This section sets out the conceptual basis and definitions that are essential to cover the successive tasks of the project related to ES. First, we set out the rationalization behind the ES cascade model as the framework that supports the definition of ES used in DRYVER, as well as to identify the biophysical agents involved in its provision. In the subsequent section, we refer to the spatial and temporal dynamics necessary to understand how the provision of ES is shaped by the environment in which they are provided, as well as how ES relate to each other. Finally, we present the ES classification system used in this report to name, define and group the different ES considered in the report.

### Ecosystem services cascade model

According to Potschin & Haines-Young, (2011), the idea of a “service cascade” can be used to summarize much of the logic that underlies the contemporary ES paradigm and key elements of its conceptual basis (Fig. 1). This cascade model shows how biophysical structures and ecological processes support ecosystem functions whose outputs are transferred as services that are defined and valued socio-economically. Specifically, biological components are involved in a large number of physicochemical cycles and biological interactions that occur within and across ecosystem boundaries, providing different ecological processes and functions simultaneously, such as productivity or recycling nutrients (Manning et al., 2018). In more detail, each ecological function has its origin in the biophysical interaction between the biological components at their different levels of organization (e.g. populations, communities or food webs) and the physical processes that control the multiple abiotic flows circulating through the landscape (e.g., flows of water, energy or matter; Kremen & Ostfeld, 2005). At the landscape level, humans get benefit from ecosystem functions in the form of a wide range of ES when the biophysical interaction occurs at the spatial scale required by the specific process (Křováková et al., 2015; Laca, 2021; Schirpke et al., 2020; Syrbe & Walz, 2012). For example, vegetation provides regulating ES such as flood mitigation or erosion protection by retaining part of the water and sediment flows in areas that drain into the river network. Similarly, provisioning ES follow the same rationalization. For example, in the pasture production ES, nutrient, water and radiation flows are used by plant organisms in defined (or undefined) socio-ecological spatial units (Schirpke et al., 2020) to generate aerial biomass that can be used for nutritional purposes.

The ES cascade model therefore attempts to capture the prevailing view that there is something of a ‘production chain’ linking ecological and biophysical structures and processes on the one hand and

elements of human well-being on the other, and that there is potentially a series of intermediate stages between them. According to Potschin & Haines-Young, (2011), this framework should also help framing a number of important questions about the relationships between people and nature, including: (1) whether there are critical levels, or stocks, of natural capital needed to sustain the flow of ecosystem services; (2) whether that capital can be restored once damaged; (3) what the limits to the supply of ES are in different situations; and (4) how we value the contributions that ES make to human well-being.



**Figure 1.** The ES cascade model initially proposed in Haines-Young and Potschin (2010) modified to separate benefits and values in De Groot et al. (2010). Unmodified figure obtained from Potschin & Haines-Young (2011).

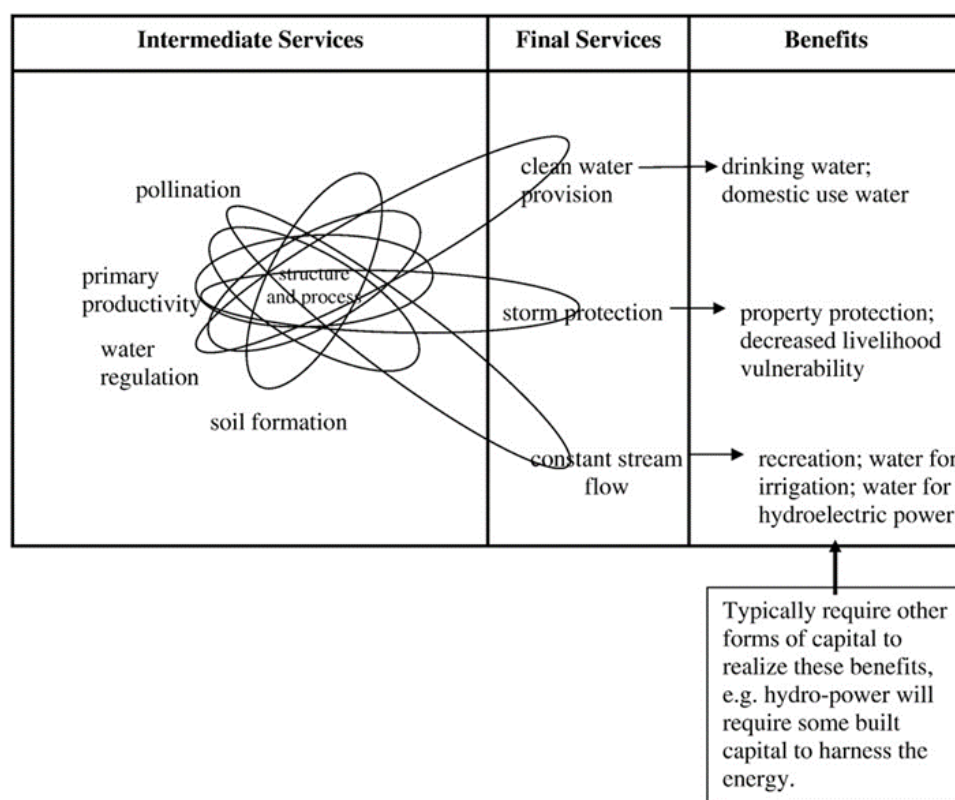
## Intermediate and final ecosystem services

One of the most widely and currently used definition of ES is the one provided by Fisher et al., (2009) which defines them as: “the aspects of ecosystems utilized (actively or passively) to produce human well-being”. According to this definition ES:

- 1) must be ecological phenomena (although some authors also include some landforms and other physical structures that appear naturally in the landscape; e.g. Keesstra et al., 2018).
- 2) do not necessarily have to be directly utilized by society, but it is essential that there is a benefit for society. In this sense, the functions, biophysical structures and processes become services if there are humans that benefit from them. Without human beneficiaries, they are not services.

In this sense, the different biological components of the ecosystem could be thought of as an ES because they provide the platform from which ecological processes and subsequent ecosystem functions occur. How much of an ecosystem process rate or which is the state of an ecosystem component required to provide an array of ES in a given environmental setting is still a pending research question (Kremen & Ostfeld, 2005; Turner et al., 1998). Clearly, some minimum level of an ecosystem structure (e.g., a density of a key specie or a given ecological process rate) is required for ‘healthy’ functioning and ES provision. This ecosystem structure has value in the sense that its prior existence and maintenance is necessary for ES provision, and is therefore a service in itself (Turner et al., 1998). This does not mean that ecosystem structure, function, and ES are identical or synonymous.

Ecosystem structure and function have been identified and studied for years, with no reference to ES provisioning to humans. So, while most ecosystem structures and processes do provide ES they are not the same thing. Following this basis, ES are usually classified in final and intermediate ES depending on the benefit provided to society (Fig. 2). Final ES, also known as Final Ecosystem Goods and Services (FEGS; Nahlik et al., 2012; Saarikoski et al., 2015), benefit people explicitly and can be accounted for in biophysical or monetary terms through measures such as fisheries output per season (provisioning ES) or number of lake visitors per year (cultural ES). In contrast, intermediate ES are often described as those ES with little or no direct benefits to people and consist of the biophysical structures and processes that maintain ecosystems in a favorable state for the provision of final ES (corresponding roughly to the regulating and supporting ES categories; [MEA] Millennium Ecosystem Assessment, 2005).



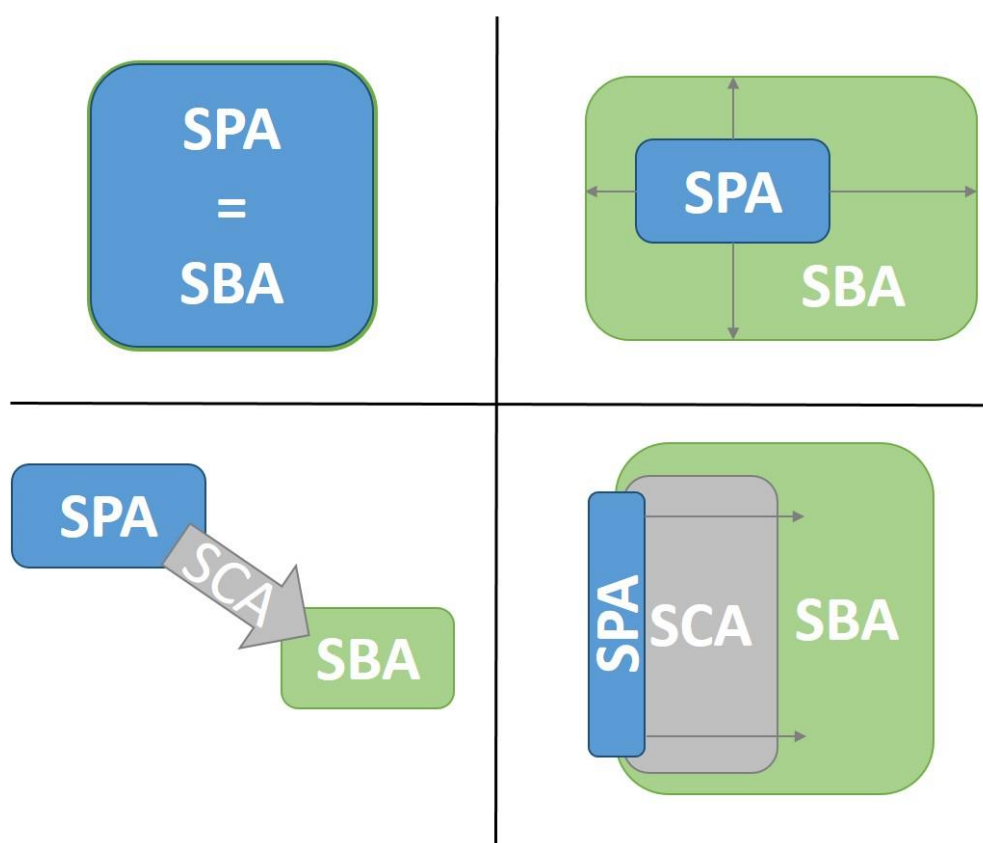
**Figure 2.** Conceptual diagram showing relationships between intermediate and final services. Joint products (benefits) can stem from individual services. Intermediate services can stem from complex interactions between ecosystem structure and processes and lead to final services, which in combination with other forms of capital provide human welfare benefits. Unmodified figure obtained from Fisher et al. (2009).

Consequently, this definition presents contradictions with the service cascade model: “not only ecosystem functions can produce an ES, but also the processes, functions and structure of the ecosystem can derive a service to the extent that they are consumed or used directly or indirectly by humanity”. However, Saarikoski et al. (2015) provided a link between the intermediate ES concept and the ES cascade conceptual framework. Following Saarikoski et al. (2015), intermediate ES are ecosystem structures, processes, and functions that support final ES provision given a particular social context. The intermediate ES term can then be seen to supersede ambiguity over the term ecosystem function and carry a clearer meaning in policy and management contexts to convey the important (yet indirect) contributions of ecosystem functions to human well-being. In an ecosystem-planning context, the intermediate ES with key contributions to final ES can then be identified, prioritized, and communicated to the public according to their level of importance for contributing to final ES.

## Spatial and temporal dynamics of ecosystem services

ES are usually provided within process-related landscape units such as watersheds, specific habitats, or natural units (i.e. functional units *sensu* Křováková et al., 2015; Laca, 2021). An advantage of the ES approach is that it shows the conditions under which nature creates benefits. However, the areas that provide ES might differ from those areas in which society benefits from these services. In this sense, we can differentiate three types of areas in the landscape in relation to the ES flows (Fig. 3; Syrbe & Walz, 2012):

- **Service-providing areas (SPA):** spatial units that are the sources of ES in a given landscape. Areas where the biophysical interaction lead by ecosystems occur to generate the ES.
- **Service-connecting areas (SCA):** spatial units connecting providing areas with benefiting areas in a given landscape.
- **Service-benefiting areas (SBA):** spatial units where the benefits from ES are required/consumed in a given landscape. Locations where ES are delivered to society.



**Figure 3.** Conceptual diagram showing possible spatial relationships between service providing area (SPA) and service benefiting area (SBA; according to Fisher et al., 2009). On the upper left 'in situ' situation: SPA and SBA are identical, i.e. the service is provided and benefits realized in the same area. On the upper right 'omni directional' situation: SBA extends SPA without any directional bias. On the lower left 'directional' – slope dependent situation: SBA lies downslope (downstream) from SPA, i.e. the service is realized by gravitational processes (cold air, water, avalanche, landslide). On the lower right 'directional' – without strong slope dependence situation: SBA lies 'behind' the SPA relating to higher-ranking directional effects. Adapted figure obtained from Syrbe & Walz (2012).

On the other hand, the ES provided by a given biological component may also fluctuate over time (Rau et al., 2018). Sometimes this variation is due to changes in abiotic and biotic fluxes within the functional unit that impact on the service-generating biophysical interaction (e.g. during the winter period, lower incident radiation leads to reductions in plant productivity and thus in pasture provisioning ES). On other occasions, the change in ES provision is determined by changes in demand (e.g. increased

demand for water provision during the summer months in tourist areas where the population is concentrated during holiday periods). Finally, in certain ES there is a time lag between the generation of the ES and its final delivery. This occurs more frequently in ES with a directional spatial relationship between the SPA and SBA areas. For example, in the drought risk mitigation ES, aquifer recharge occurs during the rainy season, but its benefit is mostly generated during drought periods.

## Relationships between ecosystem services

Recent research explores the spatial patterns of provision of multiple ES across landscapes, focusing on spatial overlap among ES provisioning as evidence of win-win opportunities for conservation of multiple ES and biodiversity (e.g. Chan et al., 2006; Egoh et al., 2008; Naidoo et al., 2008; Nelson et al., 2009). The results of these studies show there are important relationships among ES, even if the authors have not explicitly been looking for such. That is, some ES often appear together on the landscape while others seem to cancel each other. These relationships among ES are mainly caused by two mechanisms (Bennett et al., 2009). On one hand, multiple ES respond to the same driver (e.g. land uses, precipitation, etc.). Consequently, changes in a specific driver may lead to simultaneous changes (but not necessarily in the same direction) in the provision of some ES related to this driver. For example, increasing fertilizer use to improve crop production can have a significant negative effect on local provision of clean water in addition to the intended effect of increasing crop yields. On the other hand, interactions among ES themselves may cause direct or indirect changes in one ES to alter the provision of another. For example, afforestation enhances carbon sequestration, but the process of tree growth increases evapotranspiration, decreasing water availability (Pérez-Silos et al., 2021). In this sense, relationships of ES pairs can be categorized into the following three situations:

- **Synergies:** situations in which both ES either increase or decrease. For example, a synergistic relationship exists among erosion protection and flood risk mitigation ES. The roots of forest vegetation increase soil consistency, reducing sediment production, while increasing infiltration and reducing runoff that favors flood events (Pérez-Silos et al., 2021).
- **Trade-offs:** situations in which one ES increases and another one decreases. For example, water quality and agricultural production are a well-known trade-off due to differing responses to the addition of nutrients to the agricultural landscape (Carpenter et al., 1998).
- **No-effect:** situations in which there is no interaction or no influence between two ES. For example, the presence of a riparian forest buffer has positive effects on the thermal regulation of rivers, with negligible negative effects on the production of adjacent agricultural fields (Pérez-Silos, 2021; Pérez-Silos et al., 2019).

These relationships between ES often follow non-linear trajectories that vary according to spatial and temporal scale (Lee & Lautenbach, 2016; Lindborg et al., 2017). For example, the ability of floodplains to store surface water provides a flood risk mitigation service during the wet season (specifically, during flood events). In turn, floodplain inundation causes timely damage to the pasture provision ES provided by grasslands in these areas (i.e. trade-off relationship). However, at larger temporal scales, the relationship between both ES is synergistic. Firstly, the use of the provisioning ES and the activation of the flood regulation ES do not usually coincide in time. In addition, floods periodically fertilize these floodplain fields, increasing their productivity in the medium and long term. In conclusion, the relationships that emerge between different ES leads to the emergence of areas in the landscape with similar ensembles of ES that repeatedly appear together across space or time. Such ensembles, known as ES bundles (Raudsepp-Hearne et al., 2010), are a direct consequence of synergies and tradeoffs, and constitute unique providers of multiple ES, reflecting relevant socio-ecological subsystems.

## Classification of ecosystem services

Within this project we will use the Common International Classification of Ecosystem Services (CICES; Haines-Young & Potschin, 2013) because it is a widespread and accepted classification scheme (MAES; European Commission, 2014). One of CICES' advantages is that it contains a nested hierarchical structure (Haines-Young and Potschin, 2013). The highest level of CICES, the "Section", distinguishes between provisioning, regulating and cultural ES. The next hierarchical levels are "Division", "Group", and "Class". The analysis of this study was mainly based on the "Class" level of CICES. However, our tasks will usually require a more specific framework to make explicit some of the relationships between the ES and the ecosystem component that generates it. In these cases, we will scale down the "Class" level to a greater detail, generating a more concise definition of each ES.

## Challenges in the characterization of ecosystem services in river networks: spatio-temporal dynamics

Rivers are complex dendritic ecosystems. At catchment scale, the predominantly unidirectional flow of water, as well as the gravitational gradient, drive a continuous exchange of matter, energy and organisms between the terrestrial ecosystems and the river network across a wide variety of landforms (Bracken & Croke, 2007; Petersen, 1999; Tonkin et al., 2018). This intimate connection between fluvial and terrestrial ecosystems makes the fluvial ecosystem strongly subjected to the dynamics occurring in the terrestrial domain. Moreover, the relative isolation of rivers, their small size in comparison to the terrestrial area around them, their unidirectional and dendritic linear features, and their stochastic nature not only exacerbate their vulnerability (Perkins et al., 2010) but also help to connect the spatial flows that occur along the river network (Cid et al., 2021). Therefore, fluvial ecosystems are impacted by multiple human activities on the catchment, that modify both biotic and abiotic flows; such as land use changes, water abstractions, topographical alterations, hydraulic infrastructures, the introduction of invasive species or the effects of climate change (Dudgeon, 2019). These disturbances have the potential to produce strong effects on the river ecosystem components, functions and services (Barquín et al., 2015).

Furthermore, rivers are temporally dynamic ecosystems as discharge can vary greatly over time. Flow regime affects biotic interactions and lifecycles (Poff et al., 1997), biogeochemical cycles (Vázquez et al., 2015), physical properties of the channel (Bridge, 1993) and connectivity of the network and adjacent landforms (Garbin et al., 2019), so it is a key determinant for the provision of river ES (Jorda-Capdevila & Rodríguez-Labajos, 2015). This is particularly relevant in drying river networks (DRNs), where part of the network runs dry for a given time. DRNs are characterized by the presence of river reaches which have, at least, a dry phase (i.e. when surface water is absent) of variable duration and spatial extent in its annual cycle. In this sense, the alternation between the dry and wet periods occurring within a year and the duration, timing, frequency and magnitude of the drying events are extremely important controls on fluvial biodiversity and ecosystem processes all throughout the river network (Thibault Datry et al., 2018).

In this sense, research on the provision and valuation of ES in rivers has focused almost solely on perennial rivers and streams; in stark contrast, ES and their values in DRNs have been largely overlooked (Boulton, 2014). In the last decades, ecological knowledge of DRNs has grown greatly and scientists have a better understanding on the ubiquity and diversity of DRNs. However, how they interact with the terrestrial and groundwater ecosystems to determine the variability of DRN processes and patterns is still unresolved. Current riverine ecosystem models present limitations at considering

the influence of flow intermittence on their predictions about, for example, organic matter dynamics or aquatic assemblage composition (Thibault Datry et al., 2018). Consequently, new frameworks and approaches are necessary to reach an effective assessment and understanding of the spatio-temporal ES dynamics in these complex ecosystems.

In the following subsections we expose the general spatio-temporal dynamics that drive the ES provision in river networks. Firstly, we discuss how the spatial configuration of the catchment, as well as the relationships between its different ecosystems, determine the biophysical interactions involved in the ES provision. Secondly, we focus on how temporal dynamics affect these interactions by changing the hydrological conditions of the catchment and, consequently, the flow of the river network. In this sense, understanding and modelling patterns of ES provision in river networks will depend to a large extent on the ability to integrate the dynamics outlined below. For simplicity, this conceptual development is made considering a pristine catchment in which ecosystem processes and flows are not altered by human activities. Considering how human activities across the landscape change ES spatio-temporal dynamics are beyond the scope of this deliverable.

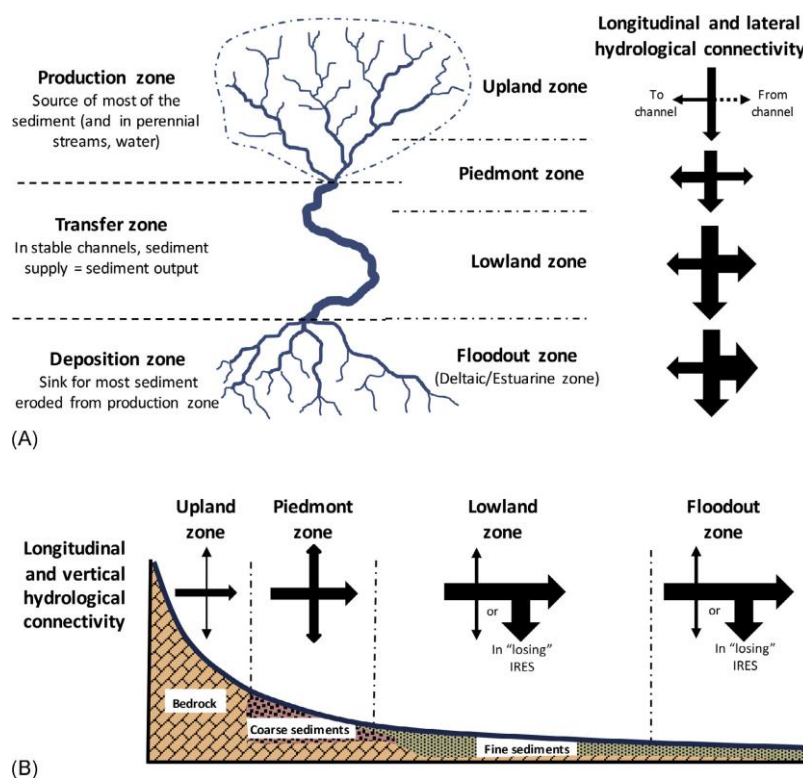
## River networks and ecosystem services: spatial dynamics

As mentioned in the previous sections, the main factors governing biophysical structures (e.g. channel morphology and substrate composition), biodiversity (e.g. microbes, invertebrates, fish, plants) and ecological processes (e.g. nutrient cycling, organic matter decomposition) in river networks are those controlling ES provision and transfer rates. Consequently, the ability of biological components to provide the different types of ES depends not only on their intrinsic features such as biodiversity or ecological conditions (Grizzetti et al., 2019), but also on its extension and location in the land/riverscape (Lamb, 2018). This is particularly relevant at a catchment scale, in which the relative position of each reach in the river network determines much of the dominant physico-chemical processes and, thus, the ES that can be potentially provided by the associated biological components (Tomscha et al., 2017). Indeed, biophysical patterns in river networks are driven by a hierarchy that controls processes from the scale of whole catchments to the scale of individual stream reaches (Townsend, 1996). Whereas climate, geology, and topography influence the catchment form and dynamics at the scale of the whole river catchment, slope, flood frequency, and substrate type better explain geomorphic and hydrologic processes within individual stream and river segments. In essence, the physical template of the catchment is configured by predictable patterns in geomorphology and hydrology, shaping highly structured river networks that change systematically from headwaters to mouth (Naiman et al., 2010; Rinaldo et al., 1993; Fig. 4).

In most natural rivers, flow and channel size increase downstream in relation to the size and flow of tributaries joining the main channel. The most basic geomorphic processes in catchments are the transportation and deposition of erosion, which also vary in relative importance along river networks. Erosive processes dominate in headwater regions. These areas are characterized by steeply sloping reaches, whose channel shape is largely bedrock-controlled and strongly influenced by local geology (Stanford & Ward, 1993). Moreover, precipitation is usually a strong driver of in-channel flow (Tooth and Nanson, 2011). Transportation dominates in the mid-reaches, whereas deposition processes dominate where the river gradient decreases and water flow is slower. Alluvial sediments and groundwater exchanges are prevalent in these regions (Stanford & Ward, 1993). This generic longitudinal pattern controls multiple eco-hydrological phenomena occurring in the river network: from sediment dynamics and landforms distribution (Wohl et al., 2015), to hydrological connectivity (Boulton et al., 2017), carbon flows (Battin et al., 2009), or even the structure and composition of stream communities (Vannote et al., 1980). The spatial variability of the different types of reaches along the river network establishes a hierarchy for the different ES potentially provided by fluvial

ecosystems depending on factors such as drainage area, river valley confinement or riparian zone extent. For example, ES that regulate sediment and surface water inputs to the river network would tend to be more important in the headwaters than in the downstream reaches (Pérez-Silos, 2021).

A catchment hierarchy acknowledges that environmental features at a given location along the river network typically reflect both local and upstream conditions (Boulton et al., 2017). In this sense, scientific understanding of how biodiversity, ecosystem processes, and ES are organized spatially across the landscape has progressed considerably with the emergence of meta-ecosystem theory (Gounand et al., 2018). This framework acknowledges the connection of different ecosystems across the landscape by means of biotic and abiotic flows. At a catchment scale, the predominantly unidirectional flow of water, as well as the gravitational gradient, drive a continuous exchange of matter, energy and organisms between the terrestrial ecosystems and the river network across a wide variety of landforms (Bracken & Croke, 2007; Cid et al., 2021; Petersen, 1999; Tonkin et al., 2018). Consequently, the hydrological connectivity between both terrestrial and fluvial ecosystems varies along the longitudinal hydro-geomorphological pattern outlined above (Fig. 4). While in headwaters the flows from terrestrial ecosystems to the river network are practically unidirectional, lateral and vertical connectivity increases in the opposite direction downstream as depositional morphologies emerge and floods become more important (Boulton et al., 2017). The river network behaves, mainly, as a receiver of the spatial flows generated in the catchment (but see in Bultman et al., 2014; Gratton & Vander Zanden, 2009; Helfield & Naiman, 2006), so its physical structuring dynamics and processes will be determined to a large extent by this intimate connection to the terrestrial matrix (e.g. Sponseller et al., 2013).

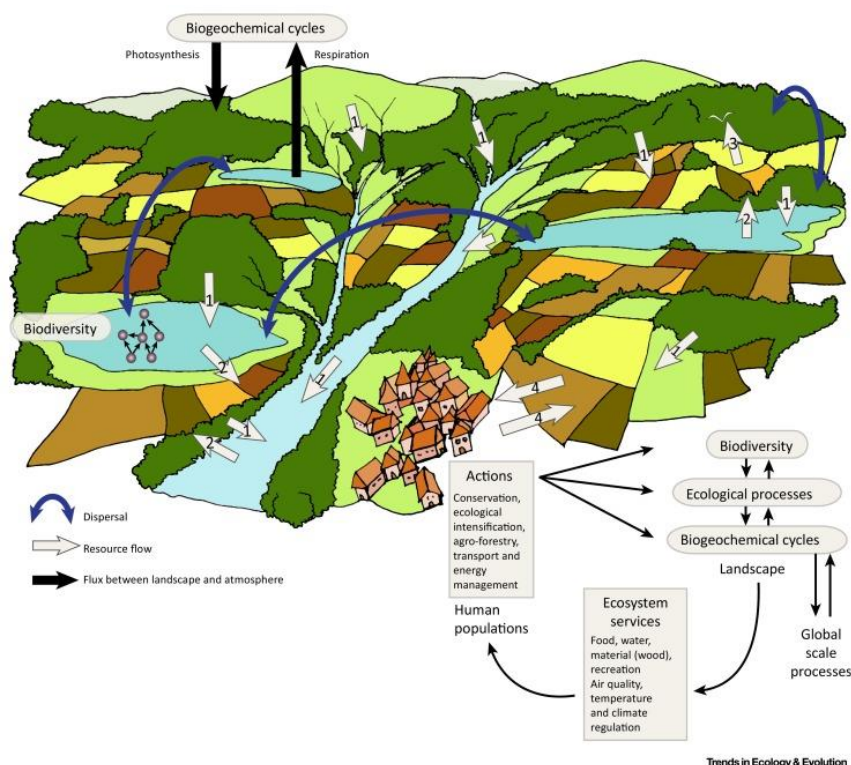


**Figure 4.** Plan (A) and transverse (B) views of geomorphological zones and their longitudinal and lateral hydrological connectivity (*black arrows*; thickness represents relative magnitudes of flow; *broken line* indicates minimal flow) in an idealized single-thread river network. In lowland and floodout zones of losing intermittent rivers and ephemeral streams (IRES), transmission losses of surface water (e.g., infiltration and recharge) are often greater represented by the large down-pointing *arrows* to the right. Unmodified figure obtained from Boulton et al. (2017).

Furthermore, river networks constrain the exchange of matter and organisms at larger spatial scales due to their dendritic topology, their temporary fragmentation by drying and the predominantly

unidirectional flow of water (Cid et al., 2021; Tonkin et al., 2018). Consequently, in river networks both local (i.e. niche selection and biotic interactions within a river reach) and regional (i.e. dispersal of organisms and spatial flows of material and energy not only across the river network but also from the landscape matrix) mechanisms interact to shape the spatial and temporal organization of populations and communities, and drive ecosystem processes and ES (Cid et al., 2021).

ES are usually provided in functional units (Křováková et al., 2015; Laca, 2021). However, as exposed earlier, the areas that provide ES might differ from those that benefit from these ES. In river networks, these spatial dynamics are particularly relevant at a catchment scale, where abiotic flows of energy, matter and water involved in the generation of ES are mediated by terrestrial and riparian ecosystems before reaching the river network (Fig. 5). For example, fluvial ecosystems have traditionally been considered as important water providers. However, few studies explicitly recognize that the water provisioning ES is primarily generated in the catchment and eventually transferred to the river network, where the transaction of benefit occurs. Incorrect identification of these ES flows can lead to poor assessment of the role of the different ecosystems involved in them and lead to ineffective solutions for ES improvement (Schröter et al., 2018). In fact, some of the ES delivered along the river network, such as hydrological regulation, are mainly provided by the terrestrial ecosystems allocated in specific areas of the catchment (Breuer et al., 2013). In those cases, the river network acts mostly as a linker (i.e. service-connecting area; SCA) between the SPA and SBA, so biological components of the fluvial ecosystems may not have much relevance to the modification of the service. However, fragmentation may ultimately impact ES provision by altering spatial flows along the river network (e.g. by dam construction or drying; respectively, Kundu et al., 2021 and Datry et al., 2018). For other ES, river networks operate simultaneously as SPA and SBA (but also as SCA if the SPA does not spatially coincide with the demand area). Although these ES are generated by biophysical structures of the fluvial ecosystem itself, some of the involved ecological processes and functions may be regulated to a greater or lesser extent by other processes and functions provided by riparian and/or terrestrial ecosystems. For example, carbon sequestration (and carbon emissions) or the chemical condition of freshwaters are ES directly linked to the activity of river organisms, and consequently derived from the structure of the river biotic communities. However, both are also controlled by other functions provided by riparian and catchment forests such as thermal buffering or hydrological processes (Bernhardt et al., 2022; Masese et al., 2017).



**Figure 5.** The conceptual diagram proposed by Gounand et al., (2018) shows the importance of the flows of matter and energy that connect different ecosystems (i.e. meta-ecosystems). At the scale of the river network, fluvial ecosystems mostly receive resource flows from terrestrial ecosystems of the catchment. These flows affect biodiversity and ecosystem processes, which themselves affect global cycles in different ways. Human populations benefit from ecosystem services provided by the landscape, and human actions conducted at the landscape scale modulate biodiversity and ecosystem functioning, and ultimately biogeochemical cycles, which in turn induce the services. Unmodified figure obtained from Gounand et al. (2018).

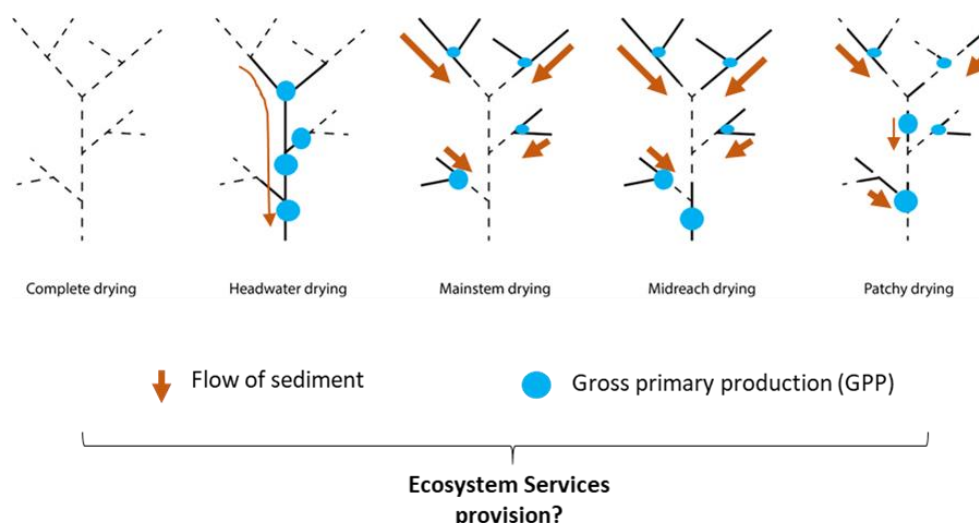
## Flow intermittence and ecosystem services: temporal dynamics

Rivers are temporally dynamic ecosystems under the control of flow variations over time (i.e. the flow regime). In DRNs, which predominate on Earth (Messenger et al., 2021), there is a dynamic cycle of three main hydrological phases: flowing, non-flowing (i.e. pools) and dry conditions (Datry et al., 2014). Over time, such cycles generate spatial patterns of flow intermittence (% of the time without water), drying duration, frequency and timing, which have been shown to strongly determine biodiversity and ecosystem functioning patterns in river networks (Crabot et al., 2020; Gauthier et al., 2020). Consequently, the duration, frequency and shift among the different hydrological phases are essential to understand the provision of ES in DRNs (Datry et al., 2018).

The three hydrological phases in DRNs respond to interactions between the multiple components of flow regime (magnitude, frequency, duration, timing, and rate of change of flow) and fluvial geomorphology (e.g., channel and floodplain shape, size, gradient, sediment composition, and location along the network; Costigan et al., 2017). Both are ultimately driven by climatic conditions such as timing and amounts of precipitation and evaporation, but also by hydrogeological features governing gains and losses of groundwater along the channel, tectonic activity (e.g., fault lines, volcanism), underlying lithology-geology (Graf & Lecce, 1988; Tooth & Nanson, 2011) and land cover. Land uses and human activities have also a high impact in wetting-drying phases (Costigan et al., 2016). All these factors modify temporally the hydrological connectivity along longitudinal, lateral, and vertical dimensions in different ways depending on the specific position in the catchment and the type of geomorphological structures present in the landscapes that river networks drain (Boulton et al., 2017). For example, during a flooding phase the water transfer channels between hill slopes, riparian zones

and river reaches in the network are dominated by direct runoff and surface-subsurface routes, while during the dry phase these pathways lose their dominant role and it is the river that contributes more water to the alluvial aquifer of the floodplain through vertical water movements through the hyporheic zone (Boulton et al., 1998).

From a meta-ecosystem perspective, flow intermittence adds a temporal variability component that has two main effects on the dynamics of ES provision. Firstly, flow intermittence governs many changes in the structure and composition of river biological communities (Boulton, 2003; Datry et al., 2014) and, consequently, affects many of the ecological functions occurring at the river reach scale (Crabot et al., 2020; Vázquez et al., 2015). According to the “cascade model” (Potschin & Haines-Young, 2011), this association between biodiversity, ecological functions and intermittence has major implications for the provision of ES. Secondly, flow intermittence also modifies, or may even interrupt, the transfer of water-borne ES from SPAs to SBAs by disrupting hydrological connectivity along surface channels, laterally across the riparian zone and floodplain and vertically along groundwater flow paths (Datry et al., 2018). In this sense, the interaction between the spatial arrangements and temporal sequences of hydrological phases likely governs the diversity and rates of ES at a single river reach, complementing the perspective of DRN as “punctuated biogeochemical reactors” (Larned et al., 2010) for ecological functions and ES such as organic matter cycling, sediments or water transfer. Finally, the spatial pattern of network drying has also been hypothesized by some authors (Datry et al., 2018) as the main driver in the distribution of SPAs, SBAs and SCAs at a catchment scale. Consequently, different suites of ES may exist depending on the spatial arrangement of perennial and non-perennial reaches in the network (Fig. 6). For example, Datry et al. (2018) have hypothesized that, as the SPAs of most regulating ES might predominate in headwaters, river networks with non-perennial headwaters would have a lower provision of these ES than other river networks with perennial headwaters but which dry out in their lower reaches.



**Figure 6.** Different idealized spatial configurations of drying events in river networks. Dotted lines represent intermittent sections. The flow of sediments and gross primary production is also hypothesized along the river network depending on the type of drying configuration to show how the spatial arrangements of intermittence would alter ecological processes and its resultant ecosystem services. Partially adapted from Datry et al. (2016).

## Conceptual model for ES provision in river networks

The main objectives of this section are (i) to outline the methodological basis that will be used for modelling ES in DRYVER and (ii) to expose the conceptual model for the ES provision in river networks

(initially restricted to the ES being considered in DRyVER). In the first section, we set out the general conceptual and digital framework on which we base ES modelling. In the second section, we conceptualize and integrate each ES into the framework whilst showing some preliminary results to facilitate understanding of the modelling approach. Finally, in the third section, we present the conceptual model of ES provision in river networks. This conceptual model builds on some of the developments made during the two preceding steps.

## **Spatio-temporal framework for modelling and conceptualizing ES provision**

Modelling the provision of ES in river networks requires a geospatial framework capable of simultaneously considering: (i) the interactions between the different terrestrial and aquatic ecosystems, and (ii) the spatial and temporal patterns of abiotic flows in the river network and the wider landscape. In this sense, the framework developed by Pérez-Silos, (2021), which integrates the basis of meta-ecosystem theory (Gounand et al., 2018) in the ES assessment framework proposed by Fisher et al., (2009) and Syrbe & Walz, (2012), has been shown to be able to consider such functional connections (i.e. based on abiotic flows) between terrestrial and river ecosystems. DRyVER could therefore capitalise on these advances to model the spatial dynamics of ES by incorporating the temporal factor that determines the drying and wetting patterns of the river network.

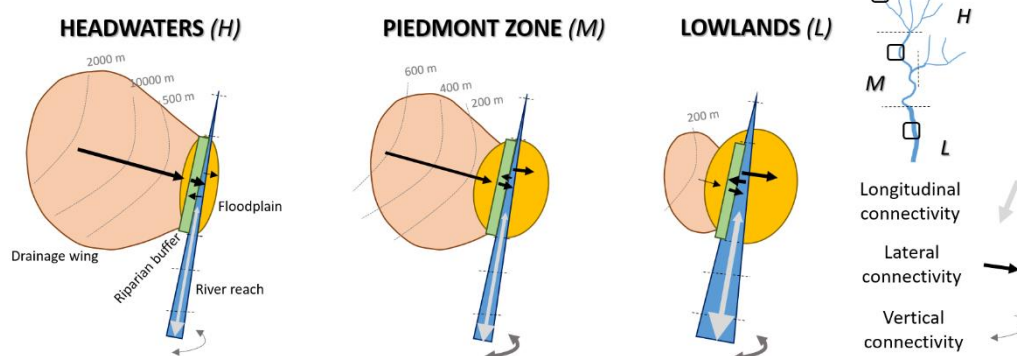
The approach developed by Pérez-Silos, (2021) considers spatial interaction between abiotic flows and some habitats in the catchment, as well as how this interaction produces benefits (i.e. ES) in areas demanded by society (i.e. SBA). Specifically, meta-ecosystem theory allows conceptualizing how material, water and thermal energy flows are driven by physical forces (e.g. water potential, gravity and radiance) that link the habitat patches in the catchment, from the upper slopes to the lower parts of the river network. These abiotic flows are modified by biological communities as they cross different patches of the landscape matrix (i.e. biophysical interaction), changing the input/output balance of resources/energy (i.e. biological function) according to the type of biological community and its location in the catchment and river network. In this sense, every biological function is produced by the biophysical interaction between the physical process that drive the abiotic flow and the habitats present within a functional unit (Křováková et al., 2015). As defined in the initial block of the document, functional units account for the spatial scale required by each biophysical interaction to potentially generate an ES. The regulation of these flows and the provision of derived resources that occurs in SPAs (i.e. source of ES) could potentially produce a benefit in society, as long as they are properly connected along the river network to the SBAs (i.e. areas of social demand for the ES generated).

According to the framework proposed by Petersen, (1999), four types of functional units, with specific hydrological functions and ecological potential within the catchment, are essential to determine the abiotic flows occurring in the river catchment. These are: drainage wings, riparian buffers, floodplains, and river reaches (Fig. 7.a). They constitute landforms not only able to incorporate the biotic-abiotic interactions at the required functional scale but also the required connectivity between different ecosystems. From a conceptual perspective, this spatial segregation makes it possible to trace the potential ES flow between the biological function generated in the terrestrial or fluvial environment (i.e. SPA) and the final ES delivered in the river network (i.e. SBA). From a modelling perspective, functional units also allow us to (i) simulate the connections between different biological components and/or landforms of the landscape, and (ii) aggregate the processes modelled from small spatial scales (e.g. a 1 m<sup>2</sup> pixel) into a spatial unit with full landscape functional meaning (Fig. 8). Consequently, in our digital framework each river reach is gravity-hydrologically connected to the terrestrial environment through these functional units, as well as, to the rest of the river network (Benda et al.,

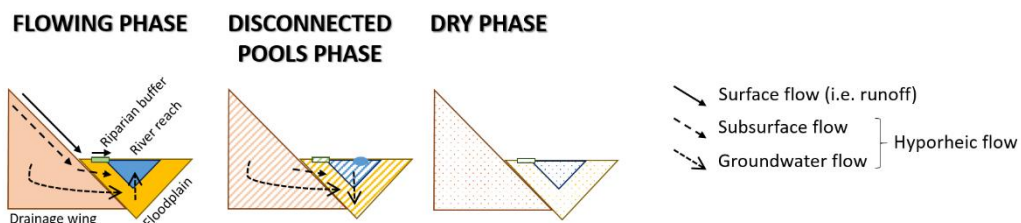
2007). While drainage wings connect unidirectionally slopes with riparian buffers and floodplains by means of different pathways (i.e. surface flow, subsurface flow and percolation), riparian buffers and floodplains are the ecotone between the terrestrial domain and the river reaches (bidirectional lateral through surface and subsurface flow, and vertical connectivity through hyporheic flow). Finally, river reaches are also bidirectionally connected to each other up or downstream (Fig. 7.a). Although most of the sediment, water and materials move downstream, we need to also consider upstream movements carried out by river organisms. River organisms movement along the river network (e.g. upstream dispersal) might alter and adds variability to the direction of the preferential downstream physical flow of mater and energy (Tonkin et al., 2018).

As mentioned above, water flow triggers and drives most of the abiotic and biotic flows in the catchment, which are therefore subject to temporal variations. Changes in catchment hydrological connectivity and river flow, especially in DRNs, lead to alterations in ecosystem connectivity and ES dynamics. In this sense, some authors have even proposed to distinguish among six hydrological phases to understand how ES are organized across DRNs in response to drying and rewetting patterns (Kaletova et al., 2021). However, it has not been demonstrated yet that 6 phases explain better ES provision than the traditional 2 or 3 phases. Furthermore, identifying, mapping and modeling 3 hydrological phases (e.g. flowing, non-flowing and dry) still represents a major challenge for river science (Allen et al., 2020; Datry et al., 2016; Leigh et al., 2016). Moreover, integrating non-flowing and/or dry phases into current monitoring approaches is to date too complex for river managers, from both hydrological and biological perspectives (Fritz et al., 2018; Magand et al., 2020; Rachel Stubbington et al., 2018). This is why drying is generally not integrated into water management plans and water legislations. Therefore, we argue that producing a conceptual model of ES provision in DRN based on the main 3 hydrological phases (i.e. flowing, non-flowing or pool and dry phases) will already be a major step forward for river scientists and managers.

**a) Functional units and their hydrological connectivity along the river network**

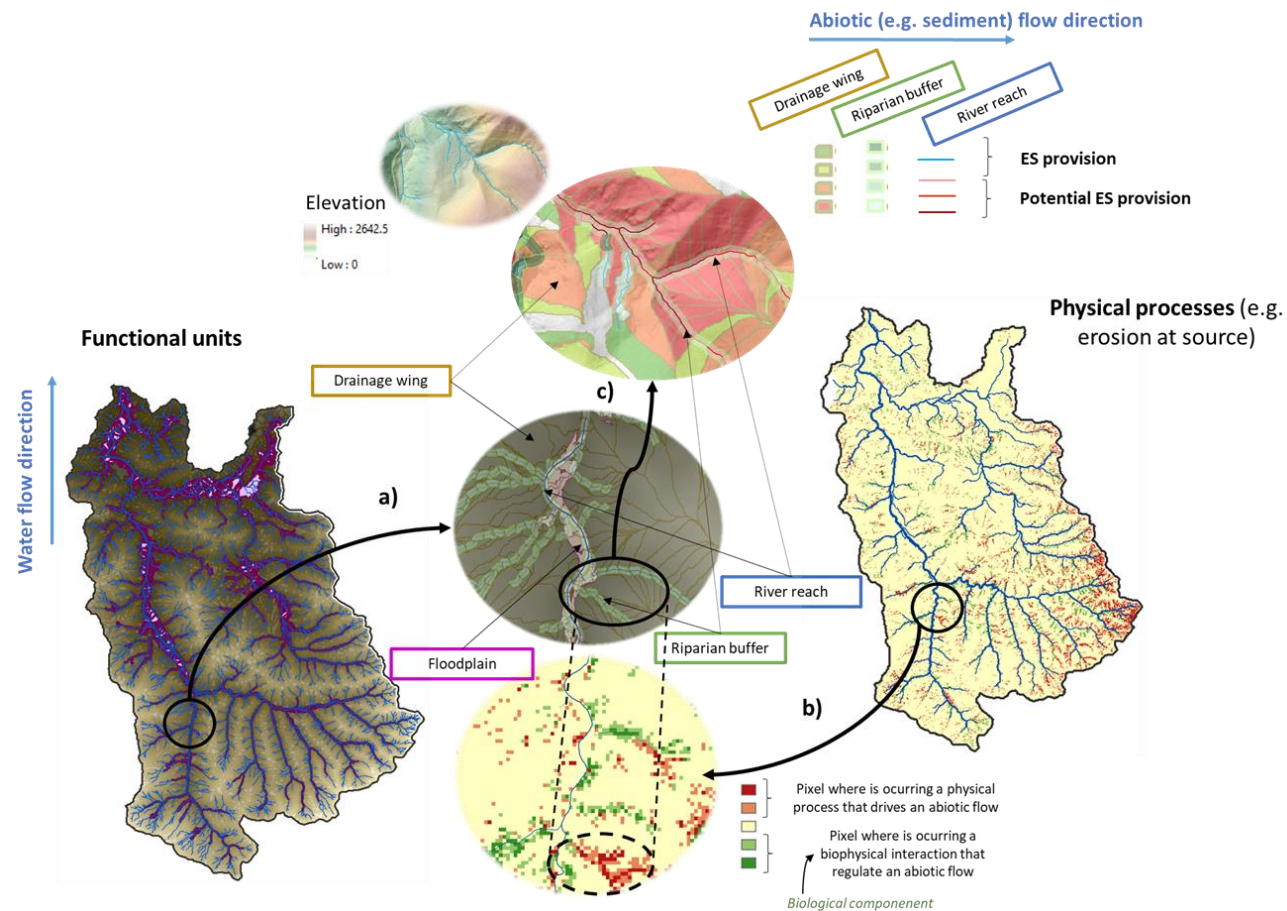


**b) Main types of water pathways connecting the functional units depending on the hydrological phase**



**Figure 7.** Plan (a) and transverse (b) views of functional units and: (a) their relative longitudinal, lateral and vertical hydrological connectivity in an idealized single-thread river network (i.e. headwaters, piedmont zone and lowlands), (b) the types of water pathways that connect the functional units depending on the hydrological phase of the DRN.

Each one of these three hydrological phases not only affect longitudinal connectivity up and downstream of the river network, but also impact lateral connectivity with the terrestrial environment (i.e. riparian buffer and floodplains) and vertical connectivity with the hyporheic environment (i.e. groundwater stored in floodplains and drainage wings). Consequently, the functional units proposed above allow us to characterize the different responses of the catchment and river network to the factors that determine the temporal variability of hydrological connectivity as well as the derived processes associated with the considered ES (e.g., sediment and nutrient dynamics). The abiotic flows of each type of functional units could be independently modified (i.e. changes in directionality and pathways of the abiotic flows; Fig. 7.b) for each of the three aquatic phases considered in DRYVER for the DRNs. Furthermore, the specific features within the functional units, as well as how they are distributed in the catchment, could be used for determining the particular spatial configuration of drying events at a DRN scale (Boulton et al., 2017; Fig. 7.b).



**Figure 8.** Digital framework for modelling and conceptualizing ES provision in river networks. (a) The river catchment is discretized into spatial units with hydrological significance: floodplains, drainage wings, riparian buffers and river reaches. (b) On the other hand, physical processes are usually modelled in raster format to take advantage of their high analytical capabilities. In this sense, the unit of analysis is the pixel. At this scale, both the physical process controlling the abiotic flux and the biophysical interaction with the biological component in case of overlap between the two are modelled. However, pixels are not able to hold information encoding complex topological relationships. (c) Functional units group sets of pixels with similar hydrological behavior (e.g. pixels that drain into the same river reach or that constitute a floodplain). In this way, by aggregating the modelled information to these functionally homogeneous units, complex relationships can be established to simulate flows of matter, energy and individuals in the catchment. This makes it possible to relate the biophysical interactions that occur in certain parts of the catchment (i.e. SPA) to the functional units that benefit from them (i.e. SBA). All this while preserving the original information (pixels) which, in a spatially explicit way, identifies at higher resolution the area where the interaction that provides the ES takes place.

## Conceptualizing the ecosystem services considered in DRyVER

Six ES were selected in DRyVER for modelling: water provisioning, flood regulation, drought regulation, erosion regulation, thermal regulation and carbon emissions. This selection was performed following a series of sequential steps which involved case study representatives. This is further described in Part 2 of this deliverable (see Regulatory Ecosystem Services section) and omitted in here for simplicity. In this sense, we select three ES directly related to WP1 (water provisioning, flood protection and drought risk mitigation), a regulatory function which is critical for instream processes modelled under WP3 (i.e. thermal regulation), other one directly related to WP3 (in-stream carbon sequestration). In relation to this later ES, rivers store carbon in riparian areas and floodplains, sediments, and dead wood while in-stream biomass is usually heterotrophic and emits CO<sub>2</sub> to the atmosphere. Flowing waters in DRyVER DRNs were always oversaturated with CO<sub>2</sub> and were carbon emitters to the atmosphere. Due to this, the in-stream carbon sequestration ES was reconceptualized and quantified as carbon emission regulation. Finally, we also included erosion regulation to further illustrate the role of terrestrial-aquatic interactions on ecosystem service provisioning on DRNs.

The following subsections conceptualize each of the ES considered in the project. In this way, their spatio-temporal dynamics are described in order to integrate them into the conceptual model proposed in section 3.3. In addition, the main variables involved in the provision of each ES are included, which will therefore be considered in the subsequent modeling process. In each sub-section we adopted the following structure:

- a. First paragraph: we briefly describe the ES.
- b. Second paragraph: we articulate the main physical processes involved in the functions related to the ES.
- c. Third paragraph (and fourth paragraph in the case of carbon emission regulation): we describe the biological components, including their respective biophysical interactions that are mainly involved in the ES provision.
- d. Fourth paragraph (fifth and sixth paragraphs in the case of carbon emission regulation): we discuss the spatio-temporal dynamics for the ES provision.
- e. Last paragraph: we summarize the spatial relationships for the ES provision (SPA, SCA and SBA; as well as potential interactions with other ES).

### Hydrological related ES

This group of ES includes those related to the hydrological response of the catchment: flood risk mitigation, drought risk mitigation and water provisioning. As defined by Park et al., (2011), we consider hydrological response as the water migration in the catchment from higher to lower hydraulic potential after a precipitation event and/or snow melting. In this sense, the path of water through the different biophysical components of the catchment determines the spatio-temporal patterns of flood and drought events in the river network. Furthermore, drying limits the amount of water available for human consumption and, consequently, it is closely linked to the quantitative component of the water provisioning ES.

The climatic pattern (precipitation and temperature) is the main factor controlling water inflows into the basin. However, the hydrological response of the catchment also depends on the hydrologic connectivity (Bracken, 2013), which it is intrinsically controlled by topo-geologic, edaphological, pedology and biologic factors (Graf & Lecce, 1988; Tooth & Nanson, 2011). These factors determine the pathways and ratios at which water entering the catchment is ultimately transferred to the river network and, as such as, the probability of flooding in the event of intense or sustained precipitation, or resilience to periods of little or no precipitation (Bracken, 2013). In this sense, flood generation is

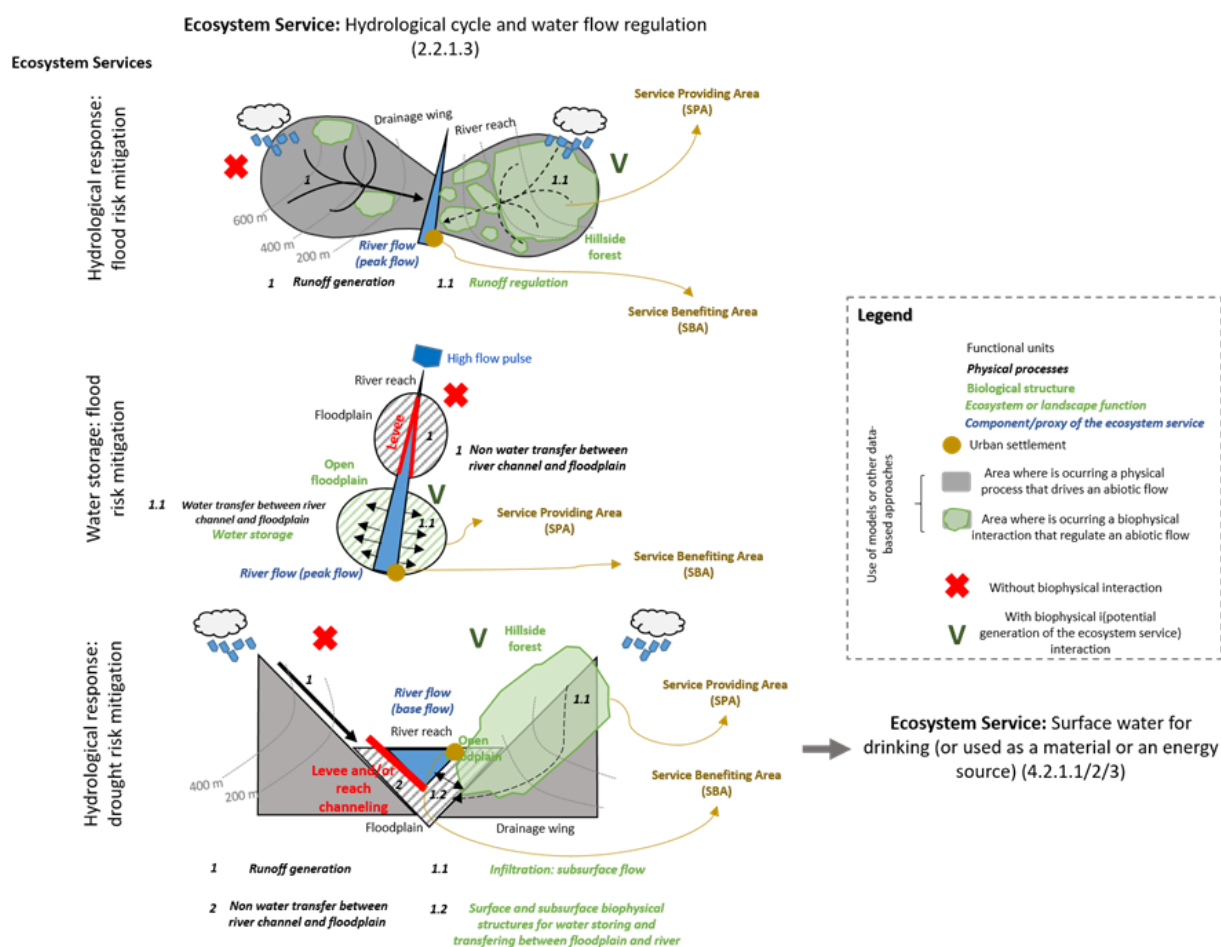
inherently linked to runoff mechanisms, while drought prevention is linked to the proportion of rainfall that can be stored in a river catchment and, subsequently, contribute to surface water flow over time (De Groot et al., 2002; Roy Haines-Young et al., 2012). In a context of low regulatory capacity, rainfall water drains rapidly from drainage wings to water bodies through surface flow. This increases the provision of water to river reaches in short term (e.g. hours or few days), but also the flood risk in case intense precipitation events. Furthermore, this usually reduces the provision of water during the dry season (Martínez-Retureta et al., 2020). On the contrary, the storage mechanisms of a catchment (e.g. aquifers, ponds, wetlands, snow-covered areas or mature forests) generate flows in periods where rainfall decreases (Mcnamara et al., 2011). Then, the existence of a close link between the regulation of the hydrological response ES and the water provisioning ES becomes obvious. Therefore, both ES depend on the storage mechanisms of the catchment, also determined by key hydrological factors such edaphological characteristics, relief and land cover (Bracken, 2013).

Biological components, and specifically the vegetal part of the ecosystems, are closely related to water resources quantity and its spatio-temporal distribution. Vegetation determines the water flow between the soil and the atmosphere through processes like interception, evapotranspiration, surface runoff, and subsurface flows. In this context, empirical evidence shows how forest improves the infiltration capacity of surface soils (Bruijnzeel, 2004; Ilstedt et al., 2007) and water retention (El Kateb et al., 2013), reducing runoff and slowing down the hydrological response of the watershed. Particularly, some studies demonstrate catchments with more than 30% mature forest have higher hydrological stability (i.e. lower peak flows and higher base flows; Belmar et al., 2018). As such, we will consider that mature forests could be the main ecosystem provider of the hydrological related ES considered in DRyVER (Fig. 9). Other relevant ecosystem component involved in the regulation of hydrological flows is the floodplain. In this regard, the lateral connection between the river and its floodplain reduce floods naturally, moderating peak flows by allowing overflow, and a way to slow down the flood wave which mitigates the risk of flooding downstream (Jacobson et al., 2015; Vis et al., 2001; Fig. 9). In this sense, vegetation, but specially riparian and floodplain forests, increases floodplain roughness, what plays a very important role for water storage during floods and for the reduction of flow peak travel time (Dadson et al., 2017; Thomas & Nisbet, 2006). Functional floodplains also support alluvial recharge during flood-postflood events, which delays the drying phase of the river network during a period of drought. By reducing flow velocity in the flooded area, the residence time of water in the floodplain increases, favoring infiltration rates and, consequently, aquifer recharge (Opperman et al., 2010; Petersen, 1999; Fig. 9).

The potential spatial pattern (independent of vegetation distribution) of provision of these ES depends primarily on the structure of the catchment and the river network. The response time to any precipitation event is primarily driven by geomorphological factors such as network length, network geology and slope gradient (Bracken & Croke, 2007; Ponce & Hawkins, 1996). In this sense, the headwaters are the areas of the catchment that accumulate a large part of the precipitation generated within the catchment, which together with steeper slopes and less infiltration due to the existence of harder materials and finer soils makes them one of the main areas for the potential provision of flood and drought regulation ES (Boulton et al., 2017). Similarly, in the valley bottoms of the intermediate sections of the river network, geomorphological structures more related to transport and sedimentation processes, such as floodplains and alluvial aquifers, are preferentially found (Naiman et al., 2010). As discussed above, these areas of the basin will also play a key role as a biophysical matrix supporting ES of flood prevention, drought regulation and, consequently, water provision. On the other hand, the infiltration capacity of a catchment varies temporally depending on the precipitation pattern and its influence on soil moisture content. For example, in highly saturated soils following continuous precipitation events, which triggers Hewlettian flow that favors runoff and (Hewlett &

Hibbert, 1967), on which the ES related to the ability of water retention by vegetation may be highly limited (Dadson et al., 2017).

Consequently, hydrological regulation ES are generated by terrestrial ecosystems in drainage wings and floodplains. Especially mature forest and floodplains influence the flow regime, and will therefore determine to a greater or lesser extent the drying and wetting pattern (i.e. hydrological phases) of the river network. In this case, the river network acts mostly as a linker (SCA) between the SPA and SBA, so biological components of the fluvial ecosystems may not have much relevance to the modification of the service (Fig. 9). However, fragmentation may ultimately impact ES provision by altering spatial flows along the river network (e.g. by dam or weir construction, water deviation or channelization; Cid et al., 2021).



**Figure 9.** Schematic outline of the meta-ecosystem approach proposed for the characterization of the hydrological related ecosystem services.

## Erosion regulation ES

Soil erosion is a process consisting of three main phases: an initial detachment of individual or mass of the soil particles from the soil, a subsequent transportation by erosion agents such as wind, water or simply gravity, and a final deposition of sediments into land depressions or delivery to water bodies (Hudson, 1964). The erosion regulation ES contemplates the mitigation of the three previous processes by means of two interrelated processes. On the one hand, the erosion regulation at source prevents soil loss and avoids the generation of sediment that can be transported (i.e. erosion regulation ES at source). On the other hand, the erosion filtering process, which retains part of the particles once the transport flow has started and reduces the amount of sediment that is finally delivered to the river

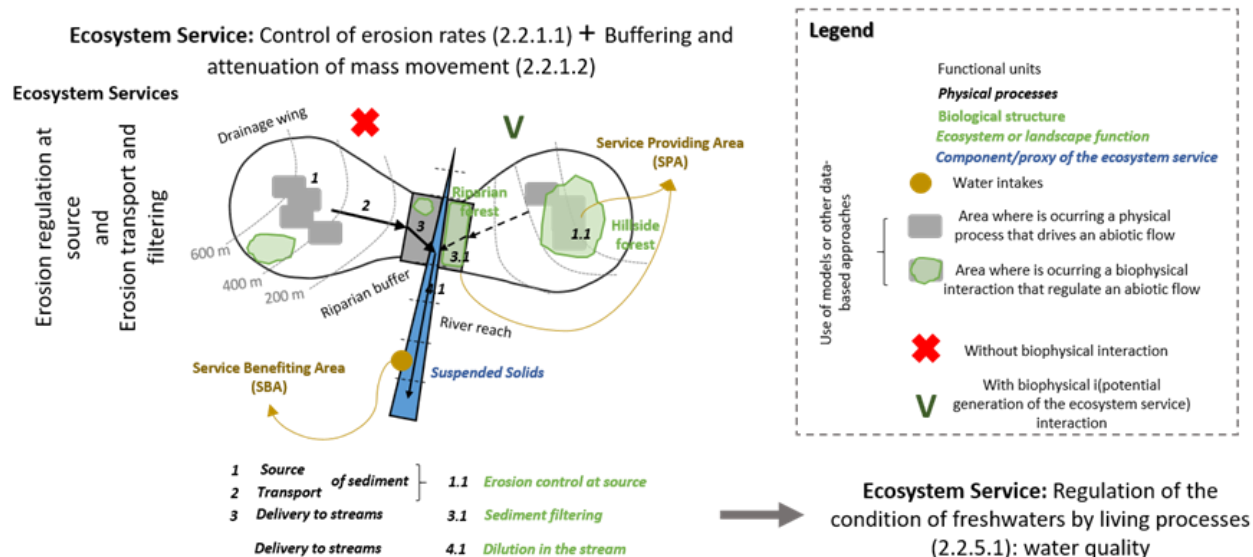
network (i.e. erosion transport and filtering ES). In this context, sediment concentration peaks in the river networks affect significantly river ecosystems (Bilotta & Brazier, 2008). During low flows, suspended solids makes water bodies more turbid and make it more difficult for light to penetrate the water column (Davies-Coelley & Smith, 2001). In addition, suspended solids have the potential of clogging the gills of aquatic organisms and covering the stream bottom that can lead to the suffocation of fish eggs and benthic macro invertebrates as well as impacting biofilm composition and functioning (Wood & Armitage, 1997). Furthermore, pesticides, some metals and other pollutants and nutrients (e.g. phosphates) may also cling to suspended sediments in water and increase the concentration of the contaminants and toxins. In high flows, high sediment concentration combined with high water flow velocity will lead to erosion and reset of benthic communities, affecting the rest of the food chain in the ecosystem (Ryan, 1991). Therefore, soil erosion also significantly influences the chemical and physical state of water, being also a process that affects the water quality ES (Keeler et al., 2012).

Sediment detachment is not exclusively the result of raindrop impacts (Hudson, 1964), but has also an origin on the overland water flow as an erosive agent (Römkens et al., 2002). Both factors generate a shear stress to the soil surface: if the soil cohesive strength is exceeded it leads to sediment detachment which can take various forms such as shallow or deep landslides, gullies, surface erosion and bankfull erosion in case of erosive action of the river on its own channel (Dietrich & Dunne, 1978; Merritt et al., 2003; however, the in-channel erosion has not been considered in DRyVER). In humid and semi-arid environments such as those in DRyVER, sediment detachment is often driven by slope steepness and slope convergence (Dietrich & Dunne, 1978). Sediment flow on the drainage wings is highly correlated with runoff routes and is directed primarily by slope (Maetens et al., 2012). If not previously retained by any biophysical structure (e.g. a depression or wetland) sediment flow is finally connected to the river network through the riparian zone.

At short timescales ( $10^1$ - $10^3$  years), vegetation is an important agent for stabilizing steep regolith-mantled slopes (i.e. erodible material). Vegetation reinforces the regolith with roots (Gabet & Dunne, 2002; Montgomery et al., 2000) and, to a smaller extent, it modifies soil moisture and subsurface hydrology through transpiration, canopy interception, redistribution of rain water, and development of preferential flow paths via live and dead root systems (Gonzalez-Ollauri & Mickovski, 2017; Hwang et al., 2015). Consequently, vegetation acts as an ES provider by preventing soil erosion in drainage wings and therefore mitigating the impact that results from the combination of the erosive power of precipitation and the biophysical conditions of a given area (Fig. 10). From an ecological point of view, more complex and mature vegetation cover contributes to greater protection of the soil (Borrelli et al., 2017). In fact, woody cover stabilizes soil surface and impedes soil movement, producing lower soil-losses in comparison to other vegetation covers (El Kateb et al., 2013; Genet et al., 2010; Marden, 2012; Wang et al., 2014). Furthermore, vegetation cover also plays a fundamental role in favoring sedimentation and, therefore, filtering sediment flows from uplands. In the catchment context, this ES of sediment filtering becomes even more important in riparian zones. In these areas, sediment is transferred from the drainage wings to the river network. The riparian area normally presents a break in the slope of the slopes, being a flatter area that favors sedimentation (Naiman & Decamps, 1997). The existence of vegetation cover would further increase the rate of sediment filtration before reaching the fluvial network (Fig. 10). According to some studies, riparian zone cover by dense tree vegetation was  $\geq 75\%$  effective in trapping sediment in agricultural areas (Lind et al., 2019) and 53% to 96% in piedmont areas (Lowrance et al., 1997; White et al., 2007). However, this effectiveness may be reduced in intense precipitation events that concentrate flow in rills and small drainage channels (rather than producing a more diffuse flow; Bereswill et al., 2013). Finally, the roots of the riparian vegetation also help to stabilize the bank of the channel, reducing channel incision and bank erosion, as well as sediment input to the river (Bigelow et al., 2012).

The control that precipitation and slope exert on the physical process of erosion, as well as their relationship with runoff, causes headwater areas to be the priority in the potential provision of the erosion regulation ES at source (Pérez-Silos, 2021). The transport of sediments from these areas to secondary tributaries often occurs through temporary channels that are activated during precipitation events (Datry et al., 2018). In both cases, the riparian areas connecting such a network structure mainly provide the potential erosion filtering ES. The temporal pattern is therefore highly decisive in the provision of erosion regulation ES. For the most part, these ES are activated during periods of precipitation that can trigger mechanical breaks and soil erosion. During the drying phases of the network, the absence of runoff and significant flows in the river network causes a loss of importance of these ES (Datry et al., 2018).

Consequently, although erosion regulation ES is generated in the drainage wings and riparian buffers, its benefit in terms of improved water quality is delivered along the river network (Fig. 10). As well as hydrological regulation ES, the river network acts as a linker (i.e. SCA) between the SPA and SBA, so biological components of the fluvial ecosystems may not have much relevance to the modification of the service. However, fragmentation may ultimately impact ES provision by altering spatial flows of the suspended solids along the river network (e.g. by dam-weir construction or embankments; Chen et al., 2010; Piton & Recking, 2016).



**Figure 10.** Schematic outline of the meta-ecosystem approach proposed for the characterization of the erosion regulation ecosystem service.

## Water temperature regulation ES

River water temperature is controlled by dynamic energy (heat) and hydrological fluxes at the air-water and water-riverbed interfaces (Hannah et al., 2008). Rivers are hierarchical systems (Montgomery, 1999) and therefore for a specific point on a river, water column temperature is determined initially by the mix of water source contributions (surface/shallow subsurface flows, groundwater, snow melt, etc.) and subsequently the energy gained or lost across the water surface and riverbed interfaces as the river flows downstream (Hannah & Garner, 2015). In this sense, the thermal regulation ES contemplates the buffering of both energy transfer processes within the river network: the direct exchange with incident solar radiation, as well as the indirect transfer from the water flows that drains to the river network. In this context, temperature regulates nearly all bio-chemical processes and affects therefore ecological processes, basic organism physiological rates, and stream community composition. The influence of stream temperature is apparent at all levels of biological organization,

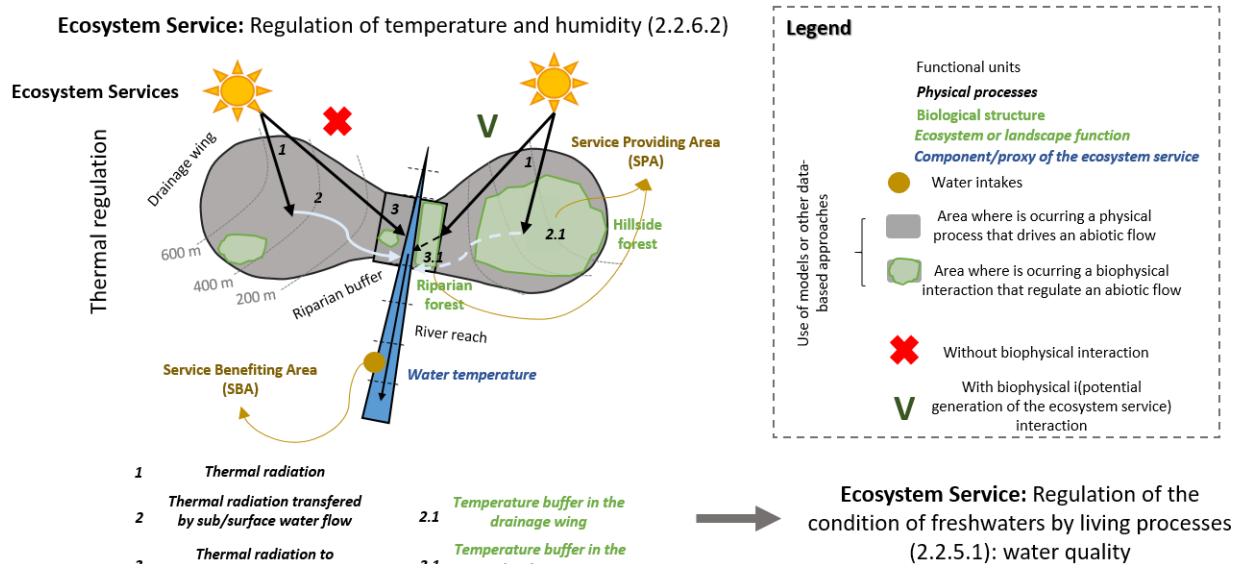
from individuals to ecosystems, and across trophic levels from primary producers (e.g. Gudmundsdottir et al., 2011), to secondary consumers (e.g. Hawkins et al., 1997), to top instream consumers (e.g., Lyons, 1996). Consequently, water temperature is an important physical property of rivers, being also an important component of the water quality ES (Keeler et al., 2012; Ozaki et al., 2003).

Heat transfer within river systems is complex, occurring through a combination of radiation, conduction, convection and advection (Webb, 1996). These energy exchanges add and remove heat to and from the river. Inputs may occur by incident short-wave (solar) and longwave (downward atmospheric) radiation, condensation, friction at the channel bed and banks, and chemical and biological processes. Losses may include reflection of solar radiation, emission of longwave (back) radiation and evaporation. In addition to these exchanges, energy may be advected by in/out-flowing channel discharge, hyporheic exchange, groundwater up/down-welling, tributary inflows and precipitation. In this sense, river temperature controls are multivariate and nested at macro-, meso- and micro-scales (Webb, 1996). According to the conceptual model proposed by Hannah & Garner, (2015), the factors that control temperature dynamics in rivers are organized in the following way. Climate drives the thermal regime in rivers and is thus the first-order control on regional patterns in the magnitude and timing of seasonal dynamics (Garner et al., 2014; Ward, 1985). Basin-wide characteristics are second-order controls; water sources, topography, geography and land use land cover may moderate the influence of climate and thus modify the timing and magnitude of subseasonal water temperature dynamics. Reach-specific controls, which interact with the water column as it moves through the catchment, such as topographic and riparian shading, hyporheic exchanges and localized groundwater contributions, may further moderate the influence of climate on water temperature dynamics. At this scale, stream flow also influences the heating capacity (volume of water) and/or cooling through mixing of water from different sources including streambed heat exchanges (Van Vliet et al., 2011).

Land cover is located at the atmosphere-soil-water interface and thus plays a key role in energy flux exchanges between these systems at meso and micro scales. On the one hand, the removal of vegetation could warm shallow aquifers in the catchment by increasing the downward heat flux from the warming land surface (Kurylyk et al., 2015). Decreased surface shading and albedo increase net radiation, leading to direct surface and subsurface heating of drainage wings (Lewis, 1998; Yoshikawa et al., 2003). In addition, the removal of tree canopy can decrease transpiration and thus increase the energy available to warm the land surface (Lewis & Wang, 1998; Rouse, 1976). Consequently, the temperature regulation ES provided by vegetation in drainage wings to shallow groundwater and subsurface flow is particularly important for the thermal regimes of groundwater-dominated streams and rivers (Fig. 11). In fact, some studies have observed an increase of about 2 °C in long-term mean annual groundwater temperature in response to deforestation (Kurylyk et al., 2015; Lewis, 1998). On the other hand, the direct shade provided by the vegetation of the riparian area to the river channel has a buffering effect on regulating stream temperatures. The removal of vegetation decreases shade, which increases solar radiation levels and consequently stream temperatures (Trimmel et al., 2018; Fig. 11). Additionally, as discussed in sect. 3.2.2, the removal of riparian vegetation can result in bank erosion. This can lead to changes in channel geometry towards a wider and shallower stream channel, all of which also increase water temperatures (Coates et al., 2005). In particular, a large number of studies have shown the effectivity of riparian forest for providing thermal regulation ES in these areas, decreasing maximum temperature and mean temperature, as well as increasing minimum temperature during the year, and reducing also diurnal variability (Malcolm et al., 2008; Mellina et al., 2002; Story et al., 2003).

At a catchment scale, the spatio-temporal patterns of potential provision of thermal regulation ES are complex due to the multiple interactions between the factors mentioned above. River water is typically cooler and less variable at subseasonal scales where water flow is sourced predominantly from groundwater (Garner et al., 2014; O'Driscoll & DeWalle, 2006; Tague et al., 2007) or snow melt (Blaen et al., 2013). Especially when the inputs come from shallow aquifers, as well as in runoff-dominated catchments or sub-catchments, the temperature of the river network is affected by the legacy of the thermal buffering provided in the drainage wings during periods of high surface warming. In this sense, not only the delay in the delivery of the ES regarding to its generation, but also the intensity of the ES provision depends on the residence time of the water at the surface or underground. Water annual temperature is generally close to the groundwater temperature at the source (e.g. in headwater streams; Benson, 1953; Scarsbrook et al., 2017), and increases thereafter with distance/stream order. This is not linear and the rate of increase is greater for small streams than for large rivers. In fact, small headwater streams, although are often groundwater dominated, can warm more rapidly than larger streams in response to catchment deforestation because their high surface to volume ratio (Caissie, 2006). Regardless of subsurface and hyporheic flow inputs, solar (shortwave) radiation input impinging on the channel, leads the increase in water temperature downstream. At smaller scales, channel orientation and shading effects of topography reduce incidental solar radiation. Consequently, the thermal regulation ES potentially provided by riparian forest is more relevant in flat or gently sloping areas with orientations that receive more cumulative radiation (i.e. south and south-west orientations in the northern hemisphere) than, for example, in incised channels with greater topographic shading (Webb & Zhang, 1997). However, above a certain threshold of channel width, the shade potentially provided by the riparian forest fails to cover a significant proportion of the channel, reducing its capacity to regulate water temperature (e.g. in the lower reaches of rivers; Pérez-Silos, 2021). In this sense, direct solar radiation is the primary factor influencing stream temperatures in summer months (Beschta et al., 1987; Sinokrot & Stefan, 1993). This fact, together with the lower flow of the river during this season, means that the thermal regulation provided by the shade of the riparian forest becomes more important during this season. The provision of this ES is even more relevant during the pool phase as the heating of water bodies is faster because the absence of flow or heat exchange in certain portions of the river network.

Consequently, thermal regulation ES is also generated in the drainage wings and riparian buffers functional units catchment, however an important part of its benefit is delivered along the river network (Fig. 11). As well as the others regulating ES exposed above, the river network acts as a linker (i.e. SCA) between the SPA and SBA. In this sense, although biological structures of the fluvial ecosystems may not have much relevance to the modification of the service, the associated riparian vegetation controls a very important part of the dynamics of this ES.



**Figure 11.** Schematic outline of the meta-ecosystem approach proposed for the characterization of the thermal regulation ecosystem service.

### Carbon sequestration and emission regulation on in-stream riverine ecosystems

Carbon (C), in its organic and inorganic forms, is a key element of the biosphere, regulating the biogeochemical processes of ecosystems. Ecosystems process, transport and transform carbon in a boundless cycle connecting all the elements of the biosphere, the atmosphere, and the lithosphere. Atmospheric inorganic carbon ( $\text{CO}_2$ ) is used up by primary producers through photosynthesis and stored as carbon in biomass and soils (Lorenz, 2013) and released back to the atmosphere by organism respiration. Therefore, ecosystems retain or release inorganic C, becoming sinks or sources of  $\text{CO}_2$ . If carbon intake is higher than release, ecosystems reduce the concentration of  $\text{CO}_2$  in the atmosphere and thus can buffer the anthropogenic effects of climate change. Therefore, carbon sequestration is considered an ES to the extent that this process helps regulating the trend experienced by the earth's climate during the Anthropocene (IPCC, 2018; Steffen et al., 2011).

However, the role of ecosystems as emitters and sinks in the global carbon cycle needs to be properly understood to optimize land management for carbon sequestration. Some studies have been performed estimating the global carbon cycling, connecting terrestrial and marine ecosystems. In those studies, rivers are considered as passive pipes that transport terrestrial carbon into the oceans. According to this view, its role in the balance of carbon exchanged between the biosphere, geosphere and hydrosphere and the atmosphere (and thus their capacity to sequester  $\text{CO}_2$ ) would be negligible. Nowadays, it has been recognized that rivers are active components of the global C cycle. They not only transport terrestrially-derived carbon to the ocean, but also store it in sediments and emit  $\text{CO}_2$  to the atmosphere (Battin et al., 2009b; Cole et al., 2007). Because of the terrestrial subsidy, river networks are simultaneously net sources of  $\text{CO}_2$  and net sinks for C in sediments. The net carbon sequestration of a whole river network depends on this balance, which responds to a complex spatio-temporal pattern that determines how the biophysical components that process carbon adapt their structures and dynamics to the geophysical world (Battin et al., 2009a).

According to Chapin et al. (2006), the term net ecosystem carbon balance (NECB) is applied to the net rate of C accumulation in ecosystems (or loss from [negative sign]). NECB represents the overall ecosystem C balance from all sources and sinks (i.e. physical, biological, and anthropogenic) which include (Fig. 12): net ecosystem exchange (the net  $\text{CO}_2$  flux from the ecosystem to the atmosphere; -NEE) + net carbon monoxide (CO) absorption + net methane ( $\text{CH}_4$ ) consumption + net volatile organic



contributing to NECB are emissions to or uptake from the atmosphere of carbon dioxide (CO<sub>2</sub>; net ecosystem exchange, or NEE), methane (CH<sub>4</sub>), carbon monoxide (CO), and volatile organic C (VOC); lateral or leaching fluxes of dissolved organic and inorganic C (DOC and DIC, respectively); and lateral or vertical movement of particulate C (PC; nongaseous, non-dissolved) by processes such as animal movement, shoot emission during fires, water and wind deposition and erosion, and anthropogenic transport or harvest. Fluxes contributing to NEP are gross primary production (GPP), autotrophic respiration (AR), and heterotrophic respiration (HR). Estimations about global carbon budget done by Regnier et al. (2013) are shown in red for the present day (2000-2010). All fluxes are in Pg C/yr, rounded to  $\pm 0.05$  Pg C/yr, and refer to total fluxes (organic and inorganic C). The stars indicate the confidence interval associated to the flux estimates, based on *The First State of the Carbon Cycle Report* (US Climate Change Science Program and the Subcommittee on Global Change Research, 2007). A black star means 95% certainty that the actual estimate is within 50% of the estimate reported; a grey star means 95% certainty that the actual value is within 100% of the estimate reported; a white star corresponds to an uncertainty greater than 100%.

According to Sutfin et al. (2016), organic carbon is stored along river networks in four primary reservoirs that contribute to reduce NEE: (i) standing riparian biomass; (ii) large downed wood; (iii) sediment, including organic matter, litter and humus on or beneath the channel surface and across the floodplain and (iv) in-stream biomass including filamentous algae, periphyton, benthic invertebrates, fish, and particulate organic matter (Fig. 13). Carbon stored values in stream biomass are frequently small compared to the other three reservoirs, which are also more persistent (Findlay et al., 2002; Naiman et al., 1987). Nevertheless, as mentioned previously, the balance between respiration and autotrophy of river biomass is the main strictly fluvial component that affects NECB, at least on short time scales. Thus, river metabolism is mainly determined by the structure of the food webs and, consequently, by the fluxes of nutrients and communities within the river network (Cid et al., 2021; Power and Dietrich, 2002). In this regard, DOM and POM are the largest carbon pool in freshwater ecosystems (Battin et al., 2009). DOM is composed of a complex mixture of allochthonous organic matter from the degradation products of terrestrial plant organic matter and autochthonous organic matter produced by the autotrophic organisms (extracellular release) or by predatory grazing, cell death and senescence and viral lysis. Most DOM and POM has shown to be of terrestrial origin, especially in headwater streams (Jaffé et al., 2013; Raymond & Bauer, 2001). This suggests that most of the compounds in OM leach directly from vegetation components (e.g., leaves, twigs, fruits; Kaplan & Newbold, 1993) or from organic matter accumulated in soils (Fiebig et al., 1990) and reach the stream via subsurface soil flow paths (Fiebig et al., 1990; Mei et al., 2012). Consequently, land cover exerts a strong influence on the origin, quantity and quality of the organic matter that constitutes the source of energy for the stream biota, this is the food resources that support an important part of the food webs of river ecosystems (Allan and Castillo, 2007). DOM and POM properties (i.e., composition and concentration) in freshwater ecosystems are strongly defined by the catchment vegetation composition (e.g., vegetation types such as herbaceous, shrub or arboreal vegetation) and the soil environment. This makes OM properties extremely sensitive to changes in land cover, as well as the way in which organic matter reaches the river network (Estévez et al., 2021).

In addition, results from a synthesis of the annual metabolic regimes of 222 rivers demonstrate that flow and light regimes are the primary controls on the timing and magnitude of river ecosystem GPP and ER (Bernhardt et al., 2022). River GPP and ER are highest where light and thermal regimes coincide and where physical disturbances are infrequent. As discussed in sections 3.2.1. and 3.2.3., both factors are highly controlled by vegetation distribution and, specifically, by the presence of forest ecosystems in both drainage wings and riparian buffers. In fact, some studies have shown how GPP/ER ratio decreases slightly in deforested headwater reaches compared to forest catchments (Rodríguez-Castillo et al., 2019). According to Bernhardt et al. (2022), for many rivers, the increased frequency of flooding and drying disturbances caused by climate and land use change may limit accumulation of autotrophic biomass and storage of organic matter in ways that reduce the availability and predictability of energy flow to support river food webs. Moreover, regarding carbon storage mechanisms in the river network, it is expected that low gradient, wide floodplains in old-growth forests with complex channel geometry (but also in headwaters; Beckman & Wohl, 2014), a high degree of lateral hydrologic connectivity

between channels and floodplains, saturated soil conditions, and slow rates of organic decay associated with relatively cold temperatures will have the greatest per unit area storage of organic carbon.

Carbon dynamics fluctuate spatially and temporally along the river network. REM has long been hypothesized to be heterotrophic ( $ER > GPP$ ) in headwaters thus contributing to carbon emissions to the atmosphere, changing toward autotrophy as we move downstream (mid-order reaches) and reverting the pattern in large rivers (Vannote et al., 1980). In this sense, headwaters receive most of the terrestrial DOC because of their drainage length, density, and inter-digitation within the landscape. Here, metabolic performance is highest because most of the microbial biomass and metabolic processes are associated with streambed surfaces, and continuous surface–subsurface exchanges ensure replenishment of nutrients, substrates and oxygen, and the removal of metabolic wastes (Battin et al., 2009). The establishment of chemical gradients along the resulting flowpaths provides niches for diverse microbial communities, providing opportunities for populations to express their physiological potential and process a broad range of organic molecules (Findlay and Sinsabaugh, 2006). Furthermore, the cross-scale interplay of the porous structures, hydrodynamic exchanges, and chemical and biological gradients in the streambed and its biofilms enhance metabolic performance in streams. However, the narrowness of the channel may result in more shading from adjacent banks and slopes, leading to less incident radiation and limiting GPP (Vannote et al., 1980). Downstream, the decrease of stable surface area per water volume explains the decreased REM (Hotchkiss et al., 2015), especially when current velocities are enough to wash out suspended algae and bacteria before they become established as active planktonic communities (Battin et al., 2009). In these sections of the river network, both the riverbed (Crump et al., 2004; Fischer et al., 2005) and the water column (Oliver and Merrick, 2006) certainly contribute to the REM, but also the hydrologic connectivity with floodplains can have a significant role (Robertson et al., 1999). The large floodplains of lowlands areas are also important stores of standing biomass as riparian vegetation due to their wet conditions and high soil productivity (Sutfin et al., 2016).

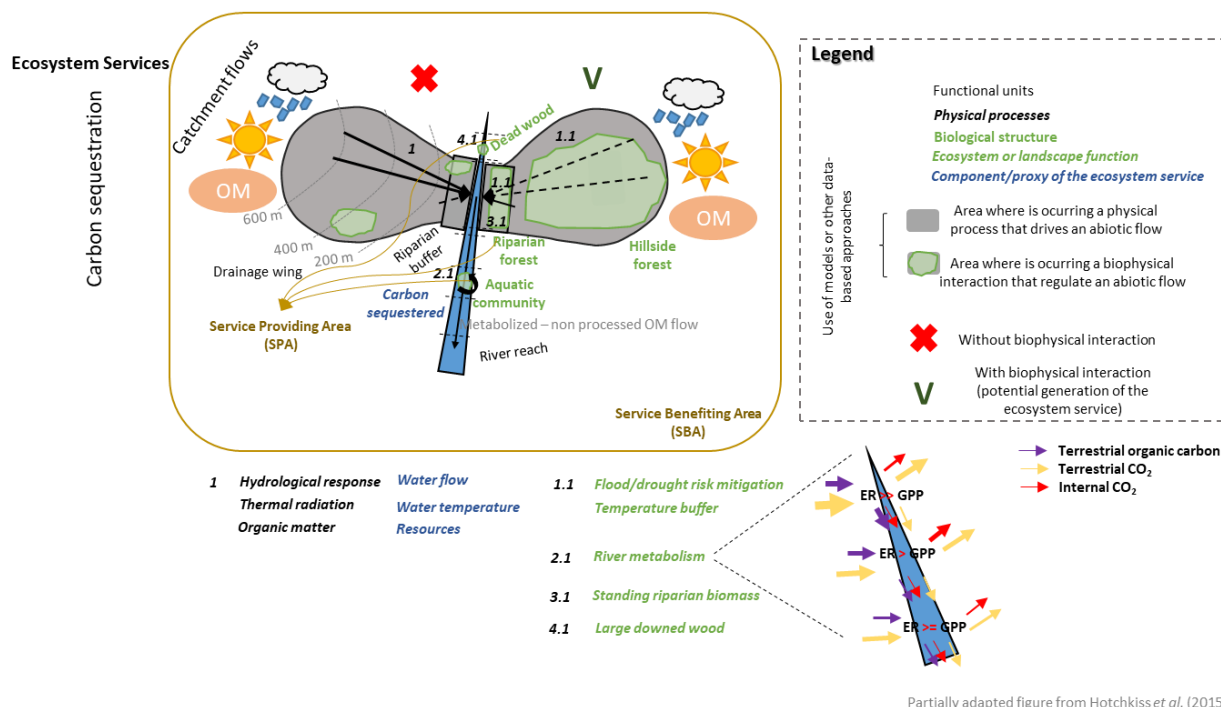
Temporal dynamics add another important dimension to the variability of REM and the way in which C is processed in river networks. During the flow phase, hydrological connectivity along the river network homogenizes the processing of DOC. On one hand, rivers are considered as net carbon sources, emitting  $CO_2$  pulses that contribute to global carbon cycles (Datry et al., 2018; Raymond et al., 2013). In addition, connectivity with drainage wings is also high, so there is an important influence of allochthonous organic matter entering the river network (von Schiller et al., 2017). In the case of flooding periods, the transport capacity of the river network is accentuated, reducing the internal processing of C and increasing the export to the sea (Cole et al., 2007). However, these flood pulses also have the capacity to activate the connection with floodplains. In this sense, storms elevate DOC exports from headwaters; flood pulses transfer this subsidy onto floodplains, where microorganisms receive another chance to transform DOC over an extended residence time (Battin, et al., 2009; Sutfin et al., 2016). On the other hand, the drying phases produce a longitudinal hydrological disconnection of the river network, which increases the spatial heterogeneity of the resources (nutrients and organic matter; von Schiller et al., 2017). Drying diminishes also lateral connections through runoff and vertical connections to the groundwater. Therefore, DOC dynamics and concentrations during drying events mostly depend on in-stream processes occurring at the stream surface (Gómez et al., 2017). In this drying phase, the ability of river networks to sequester C increases as organic matter accumulates and decomposition slows down (von Schiller et al., 2017). The cessation of flow typically leads to surface fragmentation, creating a heterogeneous landscape of isolated pools within a matrix of dry streambeds. In isolated pools, the accumulation of organic matter, high respiration rates, lack of water renewal, and weak reaeration rates commonly create hypoxic or anaerobic environments (Boulton &

Lake, 1990; von Schiller et al., 2011). Microaerophilic or anaerobic processes dominate under these circumstances (i.e. CH<sub>4</sub> emissions). The fragmentation of the river network typically evolves towards the surface stream drying. During this phase, low water availability reduces overall microbial activity through direct physiological effects, reduced diffusion of soluble substrates, and lowered microbial mobility (Humphries and Baldwin, 2003). However, stream biofilms associated with dry sediments can further process organic carbon. Specifically, heterotrophic processes associated with exoenzymatic activities are more resistant to stream desiccation than autotrophic processes (Timoner et al., 2012; Zoppini and Marxsen, 2011). Dry riverbeds can therefore emit large quantities of CO<sub>2</sub> (Marcé et al., 2019). On the other hand, the terrestrialization of river channels can also strongly influence nutrient and organic matter dynamics. In fact, the presence of terrestrial vegetation increases GPP and temporal carbon sequestration in biological structures (von Schiller et al., 2017). Finally, in some cases, increases in organic matter and nutrient concentrations after rewetting from dry conditions in river networks can cause eutrophication and potentially lead to the occurrence of hypoxic blackwater events and subsequent CH<sub>4</sub> emissions (Hladyz et al., 2011).

As a summary, the regulation of carbon emission and sequestration in rivers implies complex interactions between catchment and in-stream processes. Carbon sequestration is an ES directly linked to biophysical structures able of accumulate C in its biomass. In this sense, both the live and dead wood of riparian zones, floodplains and fluvial reaches, but also river organisms fulfil these functions. Carbon emission, however, is considered the dominant in-stream process as flowing rivers are generally oversaturated with CO<sub>2</sub>. This oversaturation is the result of the input of terrestrial inorganic CO<sub>2</sub> transported through underground waters and the respiration of terrestrial organic matter, which boost heterotrophic activity as explained previously. Rivers transform ~ 37% of the terrestrial carbon into CO<sub>2</sub>, which is emitted to the atmosphere or transported to the oceans. Due to this, river networks are considered the main source of greenhouse gas emissions to the atmosphere with an estimated carbon evasion of 1.9 – 2.3 Pg C year<sup>-1</sup> (Liu et al., 2022; Raymond et al., 2013). However, up to date these global estimates do not take into account the role of non-perennial rivers, which account for up to 60 % of the earth water courses, resulting in an underestimation of global carbon emissions (Marcé et al., 2019; Messenger et al., 2021).

In DRYVER, carbon emissions from in-stream biomass was the process considered for modelling. It is linked to REM and, consequently, to the structure of the biotic fluvial communities constitutes the SPA (Fig. 13). However, from a meta-ecosystem perspective, composition and structure of river communities, as well as, their functions (i.e. metabolism) are controlled by other functions provided by riparian and upland forests (e.g. thermal buffering or hydrological regulation) that regulate abiotic flows from terrestrial ecosystems (Bernhardt et al., 2022; Fig. 13). In the carbon emission regulation ES, changes in atmospheric conditions and subsequent climate regulation is the benefiting component. Therefore, the spatial context constituting the benefited area has multidirectional and diffuse boundaries (Fig. 13).

### Ecosystem Service: Regulation of chemical composition of atmosphere and oceans (2.2.6.1)



**Figure 13.** Schematic outline of the meta-ecosystem approach proposed for the characterization of the carbon emission regulation ecosystem service.

## Conceptual model

### How ES are potentially provided in the fluvial catchment?

The conceptual model developed for DRYvER characterizes the spatial and temporal variation in the potential provision of ES along an idealized river network, in which both the ecosystems and landforms of the catchment are in a pristine state. We considered those provisioning and regulating ES from CICES (Roy Haines-Young & Potschin, 2018) closely linked to the river network, either because they are directly produced by the fluvial ecosystem and associated ecosystems (i.e. riparian and floodplain areas), or because the benefit they generate is delivered to society in the river network itself. In this conceptual model, we exclude cultural ES as they are highly context-dependent and their variation depends not only on the biophysical properties of fluvial ecosystems, but especially on the geographical region and local sociological factors (Stubington et al., 2020). We used a meta-ecosystem approach (Gounand et al., 2018) in order to conceptualise how abiotic and biotic flows that are mediated by different ecosystems in the land/riverscape generate ES flows when the catchment is interpreted as a socio-ecological system. According to our rationalization, water acts as a vector for connectivity in the catchment (Sponseller et al., 2013). Its movement (or lack thereof) across different landforms is the main factor controlling the exchange of energy, materials and organisms in the river catchment between the ecosystems in which the biophysical interaction that generates the ES takes place (i.e. SPA) and those in which the ES is ultimately delivered (i.e. SBA). Specifically, Petersen (1999) proposed four types of spatial units, with unitary hydrological functions and similar ecological potential, that are essential to determine the abiotic flows occurring in the fluvial catchment: drainage wings, riparian buffers, floodplains and river reaches. In our conceptual model, we consider these landforms as functional units not only able to incorporate the biotic-abiotic interactions at the required functional scale to generate ES but also the hydrological connectivity between different ecosystems

along the river network (Fig. 7.a). As exposed in previous sections, both biophysical interactions and hydrological connectivity also change predictably along the river network depending on geomorphological factors (Boulton et al., 2017; Naiman et al., 2010; Rinaldo et al., 1993). Therefore, our model considers three different strata in the river network (i.e. headwaters, piedmont areas and lowlands) in which biophysical patterns are homogeneous in terms of each functional unit (Fig. 7.a). This spatial segregation of the catchment makes our model capable of delocalizing provision and delivery of ES. It allows tracing the potential ES flow between the providing functional unit, characterized by some specific abiotic and biotic conditions that determine the generation of ES, and the functional unit that receive or delivery the benefit.

In this sense, our conceptual model considers the intensity with which ES are potentially provided along the river network. For this, our model assesses the longitudinal pattern of variation in the biophysical drivers that control the generation of each ES based on the available scientific evidence (Tables 1 and 2). For example, ES such as flood and erosion protection in drainage wings, that involves the regulation of abiotic flows associated with steep slopes (e.g. sediment generation and runoff) occur mostly in headwater areas (Pérez-Silos, 2021). However, ES linked to the accumulation of flows in retention structures require depositional morphologies, with gentle slopes and more porous materials, and therefore become more important in the lower and middle reaches of the river catchment (e.g. flood protection and drought mitigation through floodplains and piedmont drainage wings; Boulton et al., 2017). As exposed in previous sections, the connectivity of these abiotic and biotic flows also varies temporally, as they depend on how water moves through the functional units of the catchment (Fig. 6.b). For example, small to medium grain materials are usually easily transported by surface runoff, but most substances mobilized by subsurface and hyporheic flows are transported as water-diluted solutes (Wohl et al., 2015). Water from groundwater flow is also cooler and more thermally stable than water entering the river network from surface flow (O'Driscoll & DeWalle, 2006; Tague et al., 2007). Both water distribution and water pathways in the catchment are ultimately driven by its recent moisture content and long-medium term storage legacy (Bracken & Croke, 2007). In this sense, our conceptual model also provides mechanisms to understand how the provision of ES varies depending on the hydrological state of the catchment (Tables 1 and 2). Thus, for each of the three hydrological phases considered (i.e. flowing, disconnected pools and dry phases, as well as flood events), our model makes explicit the changes in water pathways that occur in the functional units that generate and connect (i.e. SPA and SCA respectively) each of the ES evaluated. These variations in water pathways mainly determine the intensity and relevance of the ES during the considered hydrological phases (Tables 1 and 2). In fact, the provision of certain ES may even remain temporarily inactive in some of these phases. For example, the water temperature regulation ES is provided by the riparian forest with a higher intensity during the disconnected pools phase. The disruption of longitudinal connectivity due to loss of river flow leads to the formation of pools and non-flowing water accumulations with shallow depths and, consequently, lower specific heat. Moreover, in temperate climates, this phase usually coincides with a higher heat flow from the sun, making shade generation more relevant (Beschta et al., 1987; Sinokrot & Stefan, 1993). However, ES such as flood and erosion protection should be mostly relevant only during the flowing phase (Datry et al., 2018). In these cases, the cessation of lateral connectivity between drainage wings, floodplains and reaches during the disconnected pools and dry phases keeps these ES temporarily inactive (Nadal-Romero et al., 2008; Tuset et al., 2022).

**Table 1.** Spatio-temporal patterns for the potential provision of the **provisioning** ES considered in DRYVER. Temporal relevance in ES provision is indicated by the color of the circle (from black - highly relevant- to light grey –inactive). The light blue triangle shows the behavior of the ES during a flood event (top -provision rises-; down -provision decrease-). \* The ES has got a temporal delay and it could be generated in the previous phase. \*\* Floods favor lateral connectivity and wood deposition in storage structures. F: flowing phase; P: pool phase; D: dry phase; H: headwaters; M: piedmont; L: lowlands. Discontinuous lines show some uncertainty in our predictions or in the scientific literature.

Class CICES v5.1 (adapted from Roy Haines-Young & Potschin, 2018)	Abbreviation	Ecosystem – biological component implied	SPA: functional unit where the ES is originated	SCA: functional unit connecting supply with potential demand (pathway referred to the connection with/between river reaches)			SBA: functional unit where the ES is delivered	Spatial			Temporal		
				F	P	D		H	M	L	F	P	D
Plants (including fungi, algae), their fibres and other materials, used for nutritional, manufacturing or energy source purposes (1.1.1.1, 1.1.1.2, 1.1.1.3, 1.1.2.1, 1.1.2.2, 1.1.2.3, 1.1.5.1, 1.1.5.2, 1.1.5.3)	Vegetal biomass production (VB)	Aquatic and semi-aquatic wild plants	River reach	River and hyporheic flows	Pools – hyporheic flow	0	River reach	+			●	●	○
		Terrestrial wild plants	Floodplain – riparian buffer – river reach	Floodplain – riparian buffer	Floodplain – riparian buffer – river reach	Exposed river reach Dry river reach	Floodplain – riparian buffer – river reach	+			●	●	●
Animals, their fibres and other materials, used for nutritional, manufacturing or energy source purposes (1.1.3.1, 1.1.3.2, 1.1.3.3, 1.1.4.1, 1.1.4.2, 1.1.4.3, 1.1.6.1, 1.1.6.2, 1.1.6.3)	Animal biomass production (AB)	Aquatic and semi-aquatic wild animals	River reach	River and hyporheic flows	Pools – hyporheic flow	0	River reach	+			●	●	○
		Terrestrial wild animals	Floodplain – riparian buffer – river reach	Floodplain – riparian buffer	Floodplain – riparian buffer – river reach	Exposed river reach Dry river reach	Floodplain – riparian buffer – river reach	+			●	●	●
Genetic materials from plants, algae or fungi collected for: maintaining or establishing a population, breeding new strains or varieties, or designing and building new biological entities (1.2.1.1, 1.2.1.2, 1.2.1.3)	Vegetal genetic materials; biodiversity (VG)	Aquatic and semi-aquatic wild plants	River reach	River and hyporheic flows	Pools – hyporheic flow	0	River reach	+			●	●	○
		Terrestrial wild plants	Floodplain – riparian buffer – river reach	Floodplain – riparian buffer	Floodplain – riparian buffer – river reach	Exposed river reach Dry river reach	Floodplain – riparian buffer – river reach	+			●	●	●
Genetic materials from animals collected for: maintaining or establishing a population, breeding new strains or varieties, or designing and building new biological entities (1.2.2.1, 1.2.2.2, 1.2.2.3)	Animal genetic materials; biodiversity (AG)	Aquatic and semi-aquatic wild animals	River reach	River and hyporheic flows	Pools – hyporheic flow	0	River reach	+			●	●	○
		Terrestrial wild animals	Floodplain – riparian buffer – river reach	Floodplain – riparian buffer	Floodplain – riparian buffer – river reach	Exposed river reach Dry river reach	Floodplain – riparian buffer – river reach	+			●	●	●
Surface water for drinking, or used as a material or an energy source (4.2.1.1, 4.2.1.2, 4.2.1.3)	Surface water provisioning (SW)	Biophysical structure of the fluvial catchment	Drainage wing – floodplain	River flow	Subsurface (hyporheic) flow	0	River reach	+			▲	●	○
Groundwater (and subsurface) for drinking, or used as a material or an energy source (4.2.2.1, 4.2.2.2, 4.2.2.3)	Groundwater provisioning (GW)	Biophysical structure of the fluvial catchment	Drainage wing – floodplain	Subsurface flow	Subsurface flow	0	Drainage wing – floodplain	+			▲	●	○
Mineral substances used for nutritional, material or energy source purposes (4.3.1.1, 4.3.1.2, 4.3.1.3)	Minerals provisioning (MP)	Biophysical structure of the fluvial catchment	Drainage wing	River flow	0	0	Floodplain – riparian buffer – river reach	+			▲	●	○

**Table 2.** Spatio-temporal patterns for the potential provision of the **regulating** ES considered in DRYVER. Temporal relevance in ES provision is indicated by the color of the circle (from black - highly relevant- to light grey –inactive). The light blue triangle shows the behavior of the ES during a flood event (top -provision rises-; down -provision decrease-). \* The ES has got a temporal delay and it could be generated in the previous phase. \*\* Floods favor lateral connectivity and wood deposition in storage structures. F: flowing phase; P: pool phase; D: dry phase; H: headwaters; M: piedmont; L: lowlands. Discontinuous lines show some uncertainty in our predictions or in the scientific literature.












































































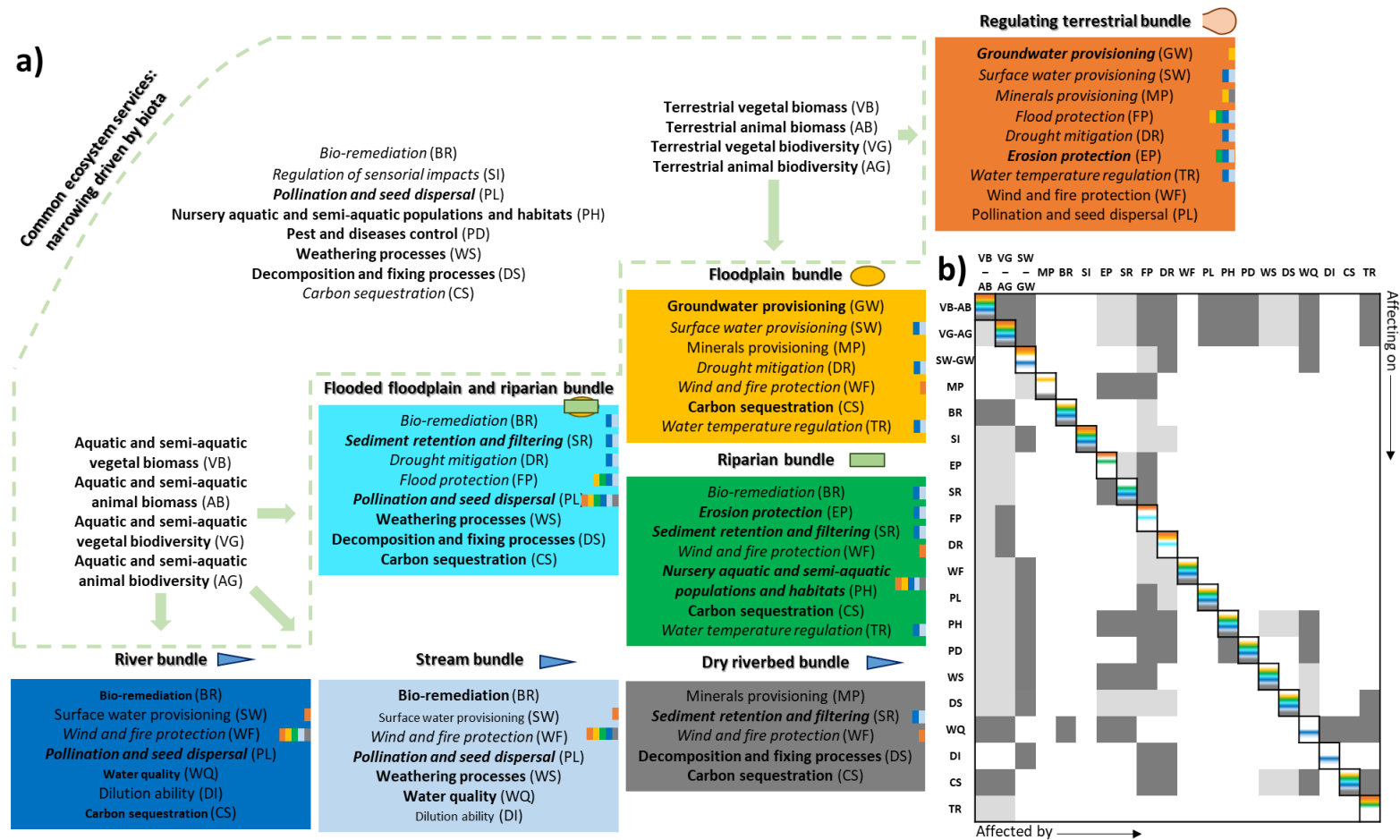
Class CICES v5.1 (adapted from Roy Haines-Young & Potschin, 2018)	Abbreviation	Ecosystem – biological component implied	SPA: functional unit where the ES is originated	SCA: functional unit connecting supply with potential demand (pathway referred to the connection with/between river reaches)			SBA: functional unit where the ES is delivered	Spatial			Temporal					
				F	P	D		H	M	L	F	P	D			
Bio-remediation/filtration/sequestration/storage/accumulation of waste, toxics and other nuisances by micro-organisms, algae, plants, and animals (2.1.1.1; 2.1.1.2)	Bio-remediation (BR)	Riparian and floodplain vegetation (i.e. riparian forest, meadows, etc.)	Floodplain – riparian buffer	River reach	**		River reach	+				**				
				Surface, subsurface (hyporheic) flows and river overflow	Subsurface (hyporheic) flow	0			-							
		Aquatic biota: macrophytes, algae and biofilm in the river channel	River reach	River reach			River and hyporheic flows	Pools – hyporheic flow	0	+						
Mediation of smell/noise/visual impacts (2.1.2.1; 2.1.2.2; 2.1.2.3)	Regulation of sensorial impacts (SI)	Riparian and floodplain vegetation, as well as animals, but specially tree cover (i.e. riparian-floodplain forest)	Floodplain – riparian buffer	Riparian buffer	-		Floodplain – riparian buffer – river reach	+								
		Aquatic biota: macrophytes, algae and biofilm in the river channel	River reach	In situ			River and hyporheic flows	0	0	-						
									+							
Mass stabilization and control of erosion rates (2.2.1.1)	Erosion protection (EP)	Catchment vegetation, but specially tree cover (i.e. hillside forest)	Drainage wing	Drainage wing			Riparian buffer	+								
				Surface and subsurface flows (and gravitational-non water)	0 (gravitational-non water)			-								
		Riparian vegetation, but specially tree cover (i.e. riparian forest)	Riparian buffer	River reach			River flow	Pools	0	+						
Buffering and attenuation of mass movement (2.2.1.2)	Sediment retention and filtering (SR)	Riparian vegetation, but specially tree cover (i.e. riparian forest)	Riparian buffer	River reach			River reach	+								
				Surface and subsurface (hyporheic) flows	Subsurface (hyporheic) flow	0		-								
		Fluvial ecosystem: macrophytes, algae, biofilm and rest of death biomass (e.g. wood) in the river channel	River reach	River reach			River and hyporheic flows	Hyporheic flow	0	+						
Hydrological cycle and water flow regulation (including flood control; 2.2.1.3)	Drought mitigation (DR)	Biological retention structures, but specially mature hillside forests and wetlands	Drainage wing	Drainage wing			River reach	+					*			
				Subsurface flow	Subsurface flow	0		-					*			
		Floodplains, favoured by the presence of wetlands and meadows	Floodplain	Floodplain			Surface and subsurface (hyporheic) flows	Subsurface (hyporheic) flow	0	+						
	Flood protection (FP)	Floodplains, specially when they are cover by floodplain forests and wetlands	Floodplain	River reach			River overflow	0	0	-						
		Catchment vegetation, but specially mature hillside forest	Drainage wing	River reach			Surface and river flows	0	0	+						
									-							
Wind and fire protection (2.2.1.4; 2.2.1.5)	Wind and fire protection (WF)	Riparian and floodplain vegetation, but specially wet forests (i.e. riparian-floodplain forest) - Fluvial ecosystem	Floodplain – riparian buffer – river reach	Floodplain – riparian buffer – river reach			Drainage wings	+								

Table 2 (continued).

Class CICES v5.1 (adapted from Roy Haines-Young & Potschin, 2018)	Abbreviation	Ecosystem – biological component implied	SPA: functional unit where the ES is originated	SCA: functional unit connecting supply with potential demand (pathway referred to the connection with/between river reaches)			SBA: functional unit where the ES is delivered	Spatial			Temporal		
				F	P	D		H	M	L	F	P	D
Pollination and seed dispersal (2.2.2.1; 2.2.2.2)	Pollination and seed dispersal (PL)	Native biota and biophysical structure of the fluvial ecosystem (water and wind)	Floodplain – riparian buffer – river reach	Floodplain – riparian buffer – river reach			Drainage wing – floodplain – riparian buffer – river reach	+	---	---	▲	●	●
				<i>River flow (i.e. hydrochory)</i>	<i>Exposed river reach</i>	<i>Dry river reach</i>							
Maintaining nursery populations and habitats (including gene pool protection; 2.2.2.3)	Nursery populations and habitats (PH)	Aquatic and semi-aquatic biota	River reach	River reach			River reach	+	—	—	▲	●	○
				<i>River flow</i>	<i>Pools – hyporheic flow</i>	0							
		Terrestrial biota	Floodplain – riparian buffer – river reach	Floodplain – riparian buffer	Floodplain – riparian buffer – river reach		Drainage wing – floodplain – riparian buffer – river reach	+	---	---	▲	●	●
				-	<i>Exposed river reach</i>	<i>Dry river reach</i>							
Pest (including invasive species) and diseases control (2.2.3.1; 2.2.3.2)	Pest and diseases control (PD)	Terrestrial and fluvial native ecosystems	Floodplain – riparian buffer – river reach	<i>In situ</i>			Floodplain – riparian buffer – river reach	+	---	---	▲	●	○
				<i>River flow</i>	<i>Pools – hyporheic flow</i>	0							
Weathering processes and their effect on soil quality (2.2.4.1)	Weathering processes (WS)	Biophysical structure of the fluvial catchment: biological (biota) and P/Q meteorization (water and wind)	Floodplain – riparian buffer – river reach	<i>In situ</i>			Floodplain – riparian buffer – river reach	+	---	---	▲	●	●
				<i>River flow</i>	<i>Pools – hyporheic flow</i>	0							
Decomposition and fixing processes and their effect on soil quality (2.2.4.2)	Decomposition and fixing processes (DS)	Aquatic, semi-aquatic and terrestrial micro biota	Floodplain – riparian buffer – river reach	<i>In situ</i>			Floodplain – riparian buffer – river reach	+	---	---	▲	●	●
				<i>River flow</i>	<i>Hyporheic flow – exposed river reach</i>	<i>Dry river reach</i>							
			River reach	River reach			Floodplain – riparian buffer	+	---	---	▲	○	○
				<i>River overflow</i>	0	0							
Regulation of the condition of freshwaters by living processes (2.2.5.1)	Water quality (WQ)	Aquatic biota of the fluvial ecosystem: animals, macrophytes, algae and biofilm in the river channel	River reach	River reach			River reach	+	---	---	▲	●	○
				<i>River flow</i>	<i>Pools – hyporheic flow</i>	0							
Dilution by freshwater and marine ecosystems (5.1.1.1)	Dilution ability (DI)	Biophysical structure of the fluvial catchment	River reach	River reach			River reach	+	---	---	▲	●	○
				<i>River flow</i>	<i>Pools</i>	0							
Regulation of chemical composition of atmosphere and oceans (2.2.6.1)	Carbon sequestration (CS)	In-stream biomass (filamentous algae, periphyton, benthic invertebrates, fish, and particulate organic matter)	River reach	River reach			Atmosphere	+	---	---	▲	●	●
				<i>River and hyporheic flows</i>	<i>Pools – hyporheic flow – exposed river reach</i>	<i>Dry river reach</i>							
		Standing riparian and floodplain biomass (i.e. specially mature forests), large downed wood and soils	Floodplain – riparian buffer	River reach			Atmosphere	+	---	---	▲	●	●
				<i>River overflow</i>	**	0							
Regulation of temperature and humidity, including ventilation and transpiration (2.2.6.2)	Water temperature regulation (TR)	Catchment vegetation, but specially tree cover (i.e. hillside forest and floodplain forest)	Drainage wing – floodplain	Drainage wing			River reach	+	---	---	▲	●	○
				<i>Surface and subsurface (hyporheic) flows</i>	<i>Subsurface (hyporheic) flow</i>	0							
		Riparian vegetation, but specially tree cover (i.e. riparian forest)	Riparian buffer	River reach			River reach	+	---	---	▲	●	○
				<i>River flow (through air by means of incidental radiation)</i>	<i>Pools – hyporheic flow (through air by means of incidental radiation)</i>	0							

The spatio-temporal patterns for the potential provision of provisioning and regulating ES in DRN (Tables 1 and 2) show that some biological components provide simultaneously multiple ES. In addition, different biological components may be associated with each other in the same functional unit in which they generate a diversity of ES (i.e. they share SPAs). This means that several ES are provided simultaneously in space (same SPA) and/or time, constituting a bundle of ES (*sensu* Raudsepp-Hearne et al., 2010). This is clearly seen on ES provided by primary producer communities (e.g., algae or macrophytes) on the “river reach” functional unit or by riparian vegetation on the “riparian buffer” functional unit. Following this, our conceptual model will consider seven ES bundles which concentrate the provision of the considered ES along the river network (Tables 1 and 2). These ES bundles are grouped under: (1) Regulating terrestrial, (2) Floodplain, (3) Riparian, (4) Dry riverbed, (5) Flooded floodplain and riparian, (6) River and (7) Stream (Fig. 14). Working with these seven ES bundles allow us not only to identify homogeneous units in the catchment for the provision of ES, but also to make explicit the dependences among ecosystems, biological components and functional units responsible of ES provisioning, delivery and benefiting areas. In this sense, our conceptual model shows how strictly and mixed terrestrial ES bundles (i.e. regulating terrestrial, floodplain and riparian ES bundles) are highly relevant in the provision of ES along the river network, as ES provided on those bundles influences the provisioning on river and stream bundles (Fig. 14a). In fact, more than the 60% of the ES provided by these ES bundles are eventually delivered in the river and stream ES bundles (Fig. 14a). Furthermore, certain of the ES provided by the regulating terrestrial, floodplain and riparian ES bundles determine the physical matrix of the fluvial ecosystem (e.g. controlling water and solid flows), affecting directly or indirectly on much of the ES provided along the river network by the strictly fluvial ES bundles (i.e. river, stream and dry river bed ES bundles; Fig. 14). For example, carbon sequestration or emission and water quality are ES provided by river and stream ES bundles. These ES are directly linked to the activity of river organisms, so they derive from the composition and structure of the fluvial biotic communities (i.e. biomass and biodiversity). However, both ES are also highly affected by other ES provided by riparian and catchment forests such as water temperature regulation or hydrological processes like flood protection or drought mitigation (Bernhardt et al., 2022; Masese et al., 2017). The riparian and regulating terrestrial ES bundles provide, respectively, these ES (Fig. 14).

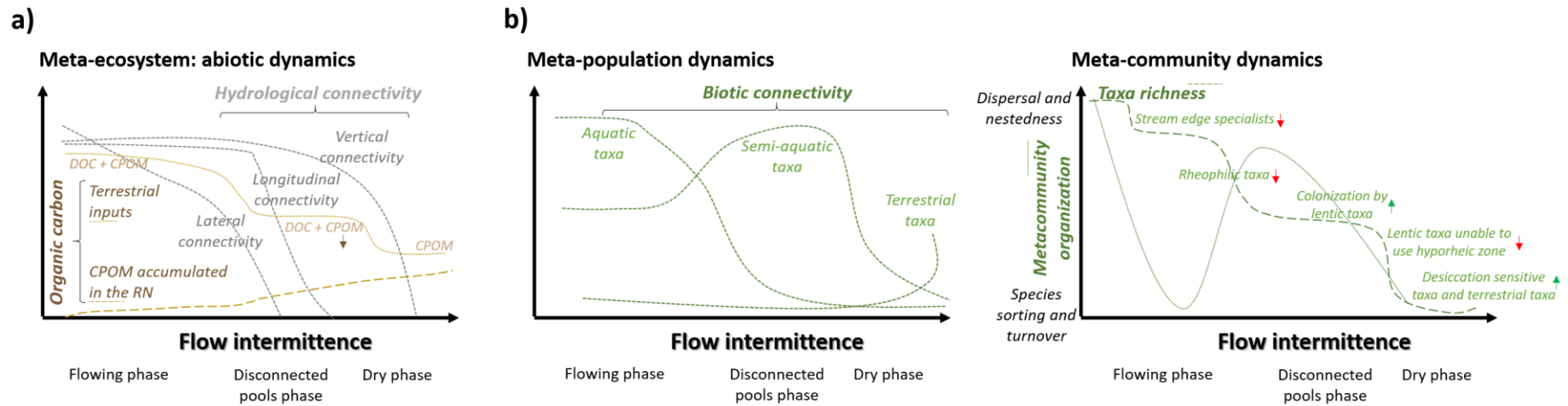


**Figure 14.** (a) The different ES bundles that interact at the catchment scale for delivering ES along the RN and associated ecosystems. ES bundles are differentiated by color. The ES that are provided and delivered by each ES bundle are listed within each ES bundle: (i) ES that are generated in the ES bundle are specified in bold, (ii) ES that are generated in the ES bundle but delivered in a different ES bundle are specified in italics (in this case, the colors on the right margin represent the ES bundle in which the ES are finally delivered), (iii) ES that are delivered in the ES bundle but have been generated in other ES bundle are specified in plain letters. Those ES provided by all the ES bundles are listed in the discontinuous box. When the provision of any of these ES is very significant in any ES bundle group, it is listed again within that ES bundle. In the case of river, stream and dry riverbed ES bundles, the letter size determines the importance in the provision of the dilution ability, carbon sequestration and water quality ES. (b) Matrix with bilateral relationships between ES (dark grey: strong effect, light grey: soft effect, white: no effect). The colors appearing in the crosses between the same ES represent the ES bundles in which the ES is generated

## How ES provision vary in different types of DRN?

When our conceptual model is evaluated at the network scale, it allows us to hypothesize how the provision of ES varies spatially and temporally between river networks with different hydrological conditions. In this sense, our conceptual model considers two types of variations in the hydrological conditions of the river network: (i) river flow over time, as well as the temporal succession between different hydrological phases and (ii) the spatial pattern of drying in the DRN. Firstly, river flow depends primarily on the climatic features of the catchment (especially the spatio-temporal pattern of precipitation, as well as other factors such as evaporation), its hydro-geomorphological configuration (e.g. lithology that may determine karstic aquifers, conditions such as altitude that determine the existence of glaciers, or terrain morphologies that determine the constriction of the network, drainage structures, etc.) and the drought mitigation ES provided by the retentive biostructures of the catchment (sec. 3.2.1; Costigan et al., 2016). These factors determine the transition time between the three different drying and wetting phases (i.e. flowing, disconnected pools and dry) of the DRN after, respectively, drought or precipitation events. Secondly, climatic features as well as hydro-geomorphological conditions also determine the drying pattern of the DRN. Our model has been conceptualized primarily for Mediterranean, temperate and continental zones, distinguishing the following common drying patterns (Boulton et al., 2017): (i) drying of headwater reaches due to cessation of rainfall and feeding of middle and lower reaches by alluvial or karstic aquifers; (ii) drying of middle reaches due to infiltration into the alluvial aquifer greater than the water supplied by the headwater drainage network, as well as river networks draining glaciers; and (iii) contraction of the river network from lower reaches. Therefore, the established framework allows to work with a wide range of DRNs: small rivers that remain dry for a long time and suffer flash flood events, rivers with short periods of drought, rivers that dry out only at the headwaters, etc.

Specifically, in the proposed conceptual model, the drying of the DRN is interpreted through several biophysical gradients that determine meta-ecosystem, meta-population and meta-community dynamics (Fig. 15). These gradients mainly explain the temporal variations in the provision of the considered ES. Firstly (Fig. 15.a), changes on hydrological connectivity modify flows among ecosystems and thus among the ES bundles of the catchment. In this sense, hydrological connectivity regulates terrestrial inputs and accumulation of organic matter along the DRN (Cid et al., 2021). These dynamics determine the resources available to the allochthonous metabolic channel of fluvial ecosystems, which supports an important part of their food webs (Allan & Castillo, 2007). Secondly (Fig. 15.b), changes in community assemblages, both alterations in the connectivity between populations and in the composition and structure of the biotic communities are also affected by changes on hydrological connectivity (Datry et al., 2017; Larned et al., 2010). Responses to the fragmentation of meta-populations cascade to altered meta-community dynamics. Reduced dispersal among isolated local communities can lead to shifts in community composition, biodiversity patterns, and biological interactions at local and regional scales (Jaeger et al., 2014), transforming ecosystem functions and services (Cid et al., 2021; Gounand et al., 2018).



**Figure 15.** Main physical (a) and biological (b) factors (in italics: hydrological connectivity, dynamics of organic matter, biotic connectivity, community organization and taxa richness) that, according to our conceptual model, mainly determine the temporal variation in the ES provision in DRNs. The response of each of the factors to flow changes in the RN along a drying gradient is hypothesized

### a.1) *Flowing phase*

The flowing phase is characterised by high levels of hydrological connectivity in all three dimensions. On the one hand, lateral (and vertical) connectivity ensures a full connection of the river network with the non-strictly fluvial ES bundles (i.e. regulating terrestrial, riparian and floodplain bundles; Fig. 16.a1). Most of the ES provided by these ES bundles regulate water, materials (i.e. sediment and dissolved or particulate organic matter) and energy (i.e. thermal energy) flows from terrestrial ecosystems, and thus are highly relevant to the functioning and services provided by the fluvial ecosystems of the river network (Fig. 14). Specifically, the regulating terrestrial ES bundle stands out in the headwaters and middle parts of the catchment, where it provides maximum values of drought mitigation, as well as, flood and erosion protection (Fig. 17). Similarly, the riparian ES bundle is also highly relevant in these sections of the river network where flows from the drainage wing and riparian functional units are dominant and the reaches have a smaller relative area (Fig. 7.a). ES provided by riparian forest such as sediment filtering, erosion protection, bioremediation or water temperature regulation stand out in this respect (Fig. 17). On the other hand, as the river reaches lower areas, the lateral connection of the river network with the drainage wings becomes less important and vertical and lateral flows with the floodplains predominate (Fig. 7.a). According to our conceptual model, the drought mitigation ES (through the recharge of shallow aquifers), as well as the provision of minerals and groundwater ES, are prominent in these piedmont zones and lowlands (Fig. 17). In addition, both the dynamics of the river network and the functioning of the fluvial ecosystem in these areas are highly determined by the effect of flooding (Junk et al., 1989). During these events, lateral connectivity becomes very important in the opposite direction to normal (i.e. from the reaches towards the riparian and floodplain functional units; Boulton et al., 2017), determining the emergence of the flooded floodplain and riparian ES bundle (Fig. 16.a1). This ES bundle works as a temporary water storage, so not only contributes to recharge floodplain aquifers and reduce flooding downstream (i.e. flood protection and drought mitigation ES), but also favors the deposition of materials and carbon storage in the retentive biostructures of these functional units (i.e. sediment retention and filtering, bioremediation, carbon sequestration, and decomposition and fixing processes for soil formation ES; Sutfin et al., 2016 and Wohl et al., 2018; Fig. 17).

During the flowing phase, both stream and river ES bundles are preponderant within the river network due to a meta-ecosystem dynamics based on full hydrological connectivity between ecosystems in the catchment and a well-represented fluvial community (Fig. 16.a1). Regarding the composition of the fluvial community, alpha diversity usually reaches its maximum during the flowing phase (Datry et al., 2014). The presence of flowing water makes possible the existence of a rich community ranging from rheophilic and desiccation-sensitive taxa to more generalist taxa (Stubbington et al., 2017). Furthermore, full longitudinal connectivity ensures biotic flows (e.g. via drift, crawling or swimming; Bogan et al., 2017) between strictly aquatic populations (Datry et al., 2014) and determines a meta-community dynamic characterized by dispersal movements (Datry et al., 2016; Fig. 16.a1). The dispersal pattern tends to concentrate movements towards the middle reaches, determining that the maximum values of alpha diversity and provision of genetic resources ES are usually located in these sectors of the river network (Stubbington et al., 2017; Figs. 17.a1 and 18). As this phase is dominated by dispersal, nestedness may be observed in the community, particularly for weak to moderate dispersers (Larned et al., 2010). In addition, the dispersive movements and the full longitudinal connectivity of this flowing phase determine that stream and river ES bundles provide a high level of pollination and seed dispersal ES in the river network (e.g. by hydrochory; Fig. 17). On the other hand, the structure of the fluvial community during the flowing phase ensures the presence of a wide variety of feeding traits, allowing food webs to use the full range of resources entering the river network. In fact, allochthonous materials (e.g. leaves and twigs) from the functional units of floodplains, drainage

wings but, above all, riparian buffers constitute a pool of detritus that forms the heterotrophic energy sources of the fluvial ecosystem (Fig. 16.a1). Specifically, specialist aquatic detritivores (i.e. shredders) play a key role in breaking this materials (coarse particulate organic matter, CPOM) to finer particles (or fine particulate organic matter, FPOM) which can be utilized by other consumers of the food webs such as filter feeders (McIntosh et al., 2017). Consequently, there is a continuous processing of the allochthonous organic matter (i.e. DOC and CPOM) entering and circulating through the river network. This, together with the input of terrestrial dissolved CO<sub>2</sub> through subsurface flow (Duvert et al., 2018), makes river network as net carbon sources during flowing phase (i.e. net heterotrophy, so low carbon sequestration ES and high CO<sub>2</sub> emissions to the atmosphere; Song et al., 2018).

However, as mentioned in previous sections, net heterotrophy, as well as other biophysical properties, vary along the longitudinal axe of the river network. This determines the differences between stream and river ES bundles and how both are distributed along the river network. On the one hand, headwaters are characterized by the presence of the stream ES bundle during flowing phase (Fig. 16.a1). Headwaters present high levels of lateral connectivity with drainage wings and riparian buffers functional units due to of their drainage length, density, and inter-digitation within the landscape (Fig. 7.a), so they receive most of the terrestrial DOC and CPOM (Leopold et al., 1964). Here, metabolic performance is highest because most of the microbial biomass and metabolic processes are associated with streambed surfaces, and continuous surface–subsurface exchanges ensure replenishment of nutrients, substrates and oxygen, and the removal of metabolic wastes (Battin et al., 2009). The narrowness of the channel results in more shading from adjacent banks and slopes, leading to less incident radiation and limiting GPP (Vannote et al., 1980). These factors determine a strong heterotrophy in this sector of the river network (i.e. low provision of carbon sequestration ES and high carbon emission), but also high provision of bio-remediation ES (Fig. 17). Furthermore, the provision of surface water and dilution capacity ES are low because the cumulative effects of the catchment are still small. However, mainly oligotrophic, well-oxygenated and cold waters (i.e. lower thermal inertia and high temperature regulation ES provided by the riparian ES bundle; Caissie, 2006) usually ensure high water quality ES (Fig. 17). On the other hand, lowlands are characterized by the presence of the river ES bundle during flowing phase (Fig. 16.a1). In this sense, the decrease of stable surface area per water volume leads to a decreased of metabolic performance, especially when current velocities are enough to wash out suspended algae and bacteria before they become established as active planktonic communities (Battin et al., 2009). Consequently, the respiration of organic matter subsidies from headwaters declines relative to GPP, which increases due to a higher photosynthetic activity (greater biomass of macrophytes and algae, as well as, availability of light; Vannote et al., 1980). This, together with a lower input of dissolved CO<sub>2</sub>, DOC and CPOM from terrestrial-based sources, reduces heterotrophy of the fluvial ecosystem compared to the headwaters (i.e. the provision of carbon sequestration ES rises slightly, reducing CO<sub>2</sub> emissions to the atmosphere; Hotchkiss et al., 2015; Fig. 17). Moreover, all the water and materials collected by the RN converge downstream. This increases highly the provision of surface water and dilution capacity ES, but also the amount of solutes in the river network. The latter, coupling with lower water-dissolve O<sub>2</sub> and higher water temperatures, leads to a reduction in the provision of water quality ES (Fig. 17). Finally, floods also affect to the provision of ES in both river and stream ES bundles (Fig. 16.a1). During these events, the level of organic matter processing decreases significantly because its direct transport predominates due to the higher flow rates (Cole et al., 2007; Raymond et al., 2016). In addition, lateral connectivity is also greatly augmented in the headwaters and piedmont areas, thus increasing the flow of materials from the drainage wings to the reaches. This results in a higher amount of solutes and suspended solids in the river network, temporarily negatively affecting the provision of water quality ES in it (Talbot et al., 2018).

### b) *Disconnected pools phase*

The disconnected pools phase is characterised by cessation of flow in the river network (i.e. break of longitudinal hydrological connectivity) as a result of partially-fully loss of the lateral hydrological connectivity with drainage wings and floodplains functional units. Indeed, as the river network dries out, lateral connectivity is rapidly lost due to the disruption of surface flow pathways (Fig. 7.b). In this sense, the pools that remain in the river network are mostly fed by inflows from the hyporheic flow, which may still be partially active (Fig. 7.b). As a consequence of the cessation of lateral surface hydrological connectivity, the regulating terrestrial and riparian ES bundles lose much of their relevance (Fig. 16.b). Both material flows and resources from the drainage wings and the riparian buffer functional units are lower than in the flowing phase (Gómez et al., 2017) and most of their associated ES start to become inactive. This fact is especially relevant for those ES whose SCA are runoff-dependent water flows, such as flood protection, sediment retention and filtering, or hydric weathering processes ES (Fig. 17). However, other ES from these ES bundles, but also from floodplain ES bundle, such as drought mitigation, bio-remediation, carbon sequestration and emission or water temperature regulation ES (Fig. 17) remain active mainly due to their dependence on subsurface flow pathways and/or their temporal delay (Tables 1 and 2). In fact, in the case of water temperature regulation ES provided by the riparian forest, the ES gains importance with respect to the flowing phase in those reaches with pools (Fig. 17). As these are static bodies with a smaller volume of water, their temperature depends more on the incident solar radiation and, therefore, on the shade potentially provided by the riparian forest (Caissie, 2006).

In relation to fluvial ecosystem, the loss of longitudinal hydrological connectivity fragments the RN into a mosaic of lotic, lentic and completely dry habitats (von Schiller et al., 2017). On the one hand, this implies the emergence of another ES bundle: the dry river bed ES bundle (Fig 17.b). This ES bundle provides ES that depend on the reach remaining totally or partially dry (Tables 1 and 2; Fig. 14), such as the provision of terrestrial biomass and minerals ES, sediment retention ES or decomposition and fixing processes for soil formation ES (Fig. 17). The dry river bed ES bundle begins to appear when lateral hydrological connectivity is lost within the river reach and gains importance in the river network with respect to the stream and river ES bundles as flow cessation and intermittency increase (Fig. 15.a). In this sense, disconnected pools are patches of stream and river ES bundles that provide, in an altered way, the ES that characterize them during the flowing phase (Fig. 16.b). For example, a lower surface water provisioning ES in these pools also implies a lower dilution capacity ES. Consequently, higher concentration of solutes, together with higher water temperatures and poor oxygenation rate, often leads to reductions in water quality ES compared to flowing phase (Gómez et al., 2017; Fig. 17). On the other hand, community composition and structure also vary sharply during the disconnected pools phase. Indeed, meta-population dynamics are characterized by disruption in the biotic connections between strictly fluvial populations (e.g. most of the fishes or many species of Ephemeroptera, Plecoptera and Trichoptera) due to the loss of longitudinal and lateral hydrological connectivity between pools (Boulton, 2003; Kerezszy et al., 2017). This can be very relevant in the face of gene flow disconnection, as well as increase the relevance of density-dependent processes influence trophic structure and functioning changes in densities and biomasses due to habitat contractions (e.g. resource scarcity or increased predation; Dewson et al., 2007). Consequently, during disconnected pools phase, fluvial community is dominated by lentic and semi-aquatic species, as well as, the variable presence of terrestrial species depending on the size of dry bed patches (Stubbington et al., 2017; Fig. 15.b). Regarding meta-community dynamics, the richness of the community typically declines comparing to communities present during flowing phases (Datry et al., 2014; but see in Bonada et al., 2007 and Dieterich & Anderson, 2000). In this sense, shifts from flowing to nonflowing conditions are likely to cause a rapid increase in the importance of species sorting, including adaptations to

environmental conditions that occur during flow cessation, biotic interactions within contracting pools, and predation by terrestrial organisms. When species sorting dominates, species turnover may be observed more commonly (Datry et al., 2016; Tonkin et al., 2016; Fig. 15.b). Subsequently, the relative importance of dispersal in accounting for community structure increases with the arrival of large specialist predators with strong flying abilities, such as dragonflies (Odonata), diving beetles (Coleoptera), and some true bugs (Heteroptera; Bogan & Boersma, 2012; Fig. 15.b). Relating feeding groups, the proportion of predators typically increases as flow ceases and prey become concentrated in contracting pools (Boulton & Lake, 2008). These increases are accompanied by corresponding declines in feeding modes associated with flowing phases, including filter feeding and shredding (Bogan & Lytle, 2007).

Both changes in hydrological connectivity and in the composition and structure of communities alter meta-ecosystem dynamics and the provision of other in-stream ES. On the one hand, in comparison to flowing phase, the longitudinal and lateral transport of DOC and CPOM is halted in disconnected pool phase (Boulton, 1991; Corti et al., 2011; Langhans et al., 2008). Consequently, CPOM from the riparian buffers functional units accumulates and terrestrial vegetation starts to develop in the dry bed of the reach (Boulton, 1991; Corti et al., 2011; Figs. 16.a and 17). In this sense, the accumulation of organic matter here represents a temporary carbon storage. This, together with a reduced decomposition rate of this organic carbon due to the low abundance of shredders and reduced microbial activity (Corti et al., 2011; Maamri et al., 2001), results in an increase in the carbon sequestration (thus a decrease in carbon emissions) ES compared to flowing phase (Stubbington et al., 2020; Fig. 17). Furthermore, this ES is also enhanced by the photosynthetic activity of the terrestrial vegetation that colonizes the river reach (von Schiller et al., 2017). On the other hand, both disconnected pools and dry habitats can continue to process relevant quantities of organic material during this phase due to the effects of leaching during rain events, photodegradation by ultraviolet light (Austin & Vivanco, 2006; Fellman et al., 2013), fermentation and accumulation of toxic compounds in anoxic pools (Boulton & Lake, 1990; von Schiller et al., 2011), and nutrient uptake by microbes and invertebrates in standing pools and in the hyporheic zone (Corti et al., 2011; Timoner et al., 2012). Consequently, the configuration of river, stream and dry river bed ES bundles during the disconnected pools phase can still emit relevant amounts of CO<sub>2</sub> and CH<sub>4</sub> gases that reduce slightly the total carbon sequestration ES (Fig. 17). In addition, they maintain a relatively high ratio of decomposition and fixation processes ES, as well as, a high sediment retention ES that potentially contribute to soil formation (Fig. 17).

Finally, the consequences for the provision of ES along the river network will largely depend on which sections of the network are disconnected (Fig. 16.b). This is particularly relevant because of its link to the pattern of organic matter accumulation in dry channels. On the one hand, the headwater streams are covered by higher densities of riparian forest, which usually means that a greater amount of leaf biomass ends up reaching the riverbed (Naiman et al., 1998; Vannote et al., 1980). Therefore, DRN that contract along the upper reaches have a greater potential to store CPOM (at least temporarily) than rivers with other drying patterns (Fig. 16.b1; Fig. 17). Furthermore, as mentioned previously, piedmont and lowlands reaches are less heterotrophic than headwaters, so annual balances of CO<sub>2</sub> emissions should be lower in these type of DRN. On the other hand, DRN that remain in the flowing phase at the headwaters still experience the delivery of materials and sediments from the catchment. This flow of matter is transported along the DRN and deposited downstream in non-flowing sections (very high sediment retention ES; Datry et al., 2014; Figs. 16.b2 and 16.b3; Fig. 17). However, CPOM and DOC contained in these matter and sediments have been previously processed in the flowing headwaters, so this configuration may produce higher CO<sub>2</sub> emissions than the other drying patterns.

### c) *Dry phase*

Dry phase is characterized by the continuation of the trends of the disconnected pool phase. On the one hand, regulating terrestrial, floodplain and riparian ES bundles are not relevant to the river network due to the complete breakdown of hydrological connectivity (i.e. including hyporheic flow; Fig. 15.a and Fig. 16.c). Consequently, the ES provision in these ES bundles is virtually inactive during this phase (Fig. 17). On the other hand, the disappearance of lentic habitats produces a strong change in the meta-population and meta-communities dynamics, as well as in the provision of ES in the reach functional units (Fig. 15.b and Fig. 16.c). As the drying of the riverbed progresses, biological connectivity between semi-aquatic taxa is disrupted. Desiccation-sensitive taxa remain in their refugia (e.g., the hyporheic environment; Stubbington, 2012) until they eventually disappear due to the loss of all hyporheic flow. The fluvial community undergoes a terrestrialization (Stubbington et al., 2017), characterized by species turnover related to the environmental filtering predominant during this dry phase (Datry et al., 2016). In addition, richness reaches minimum values compared to the flowing phase, and may decrease with distance upstream if headwaters are isolated and adult insect flight is the dominant recolonization mechanism (Stubbington et al., 2017).

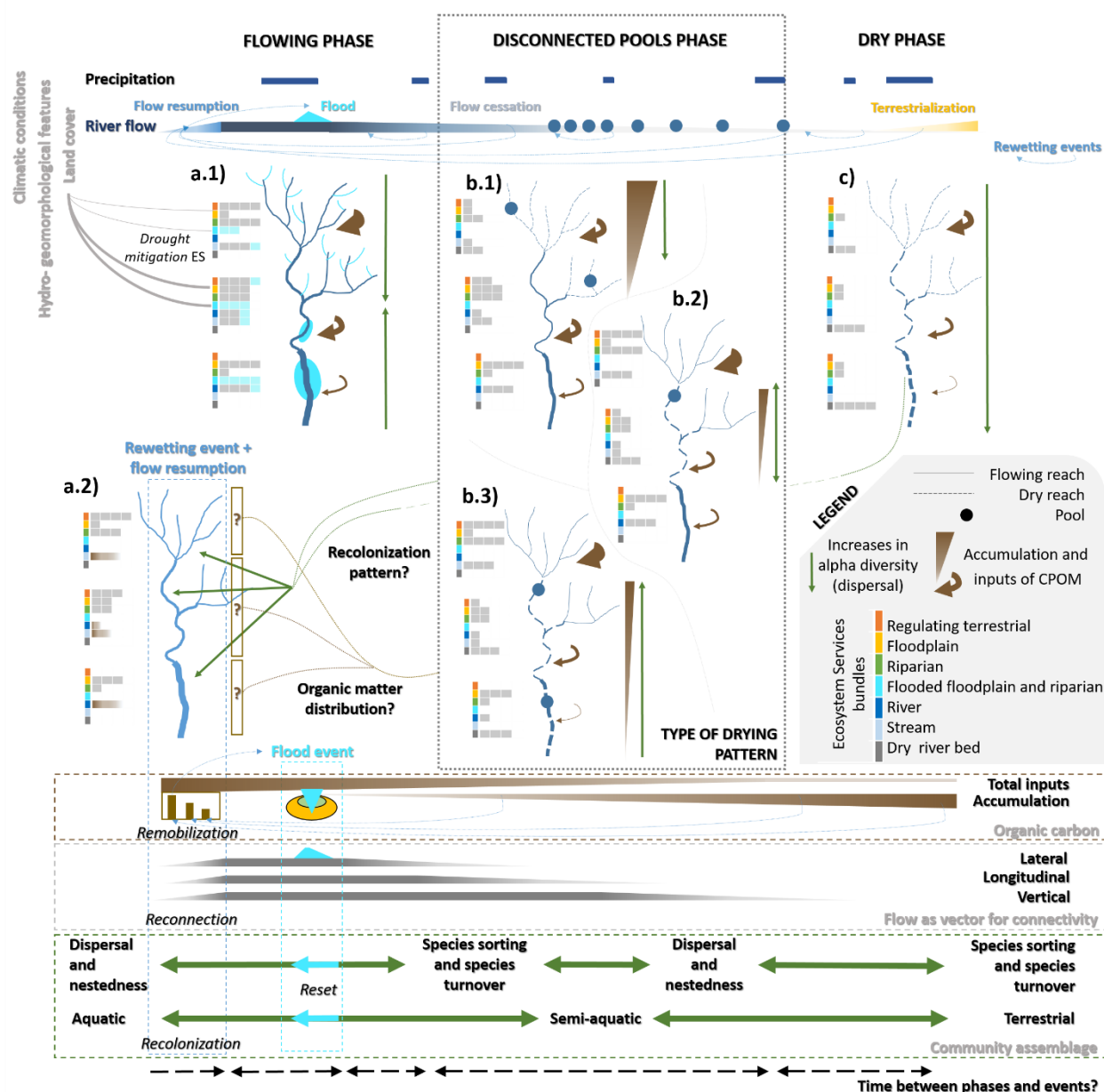
The disappearance of aquatic and semi-aquatic habitats and communities leads to the disappearance of the river and stream ES bundles, whereby reach functional units are homogenized in the dry river bed ES bundle (Fig. 16.c). This generally means an increase in the provision of their characteristic ES (Fig. 17): terrestrial biomass and biodiversity production, minerals provisioning, sediment retention, decomposition and fixing processes for soil formation and carbon sequestration. More specifically, decomposition processes are slowed by the drying of the hyporheic environment, although the terrestrialization of the community itself (e.g., biological weathering by colonizing terrestrial plants) contributes to maintaining relatively high decomposition rates for soil formation (Arce et al., 2019). In this sense, the accumulation of organic matter in the channel continues (with very low processing rates) following the pattern described above of greater accumulation in headwaters than in lowlands (Fig. 16.c and Fig. 17). The volume of this temporary organic matter storage depends on: the duration of the flow cessation period, the spatial pattern of the DRN desiccation (i.e. Figs. 16.b1, 16.b2 or 16.b3), and the timing of riparian forest defoliation. For example, if tree defoliation coincides with dry phase, a greater amount of organic matter is deposited in the reach functional units (von Schiller et al., 2015).

#### a.2) *Flowing phase (flow resumption after rewetting event)*

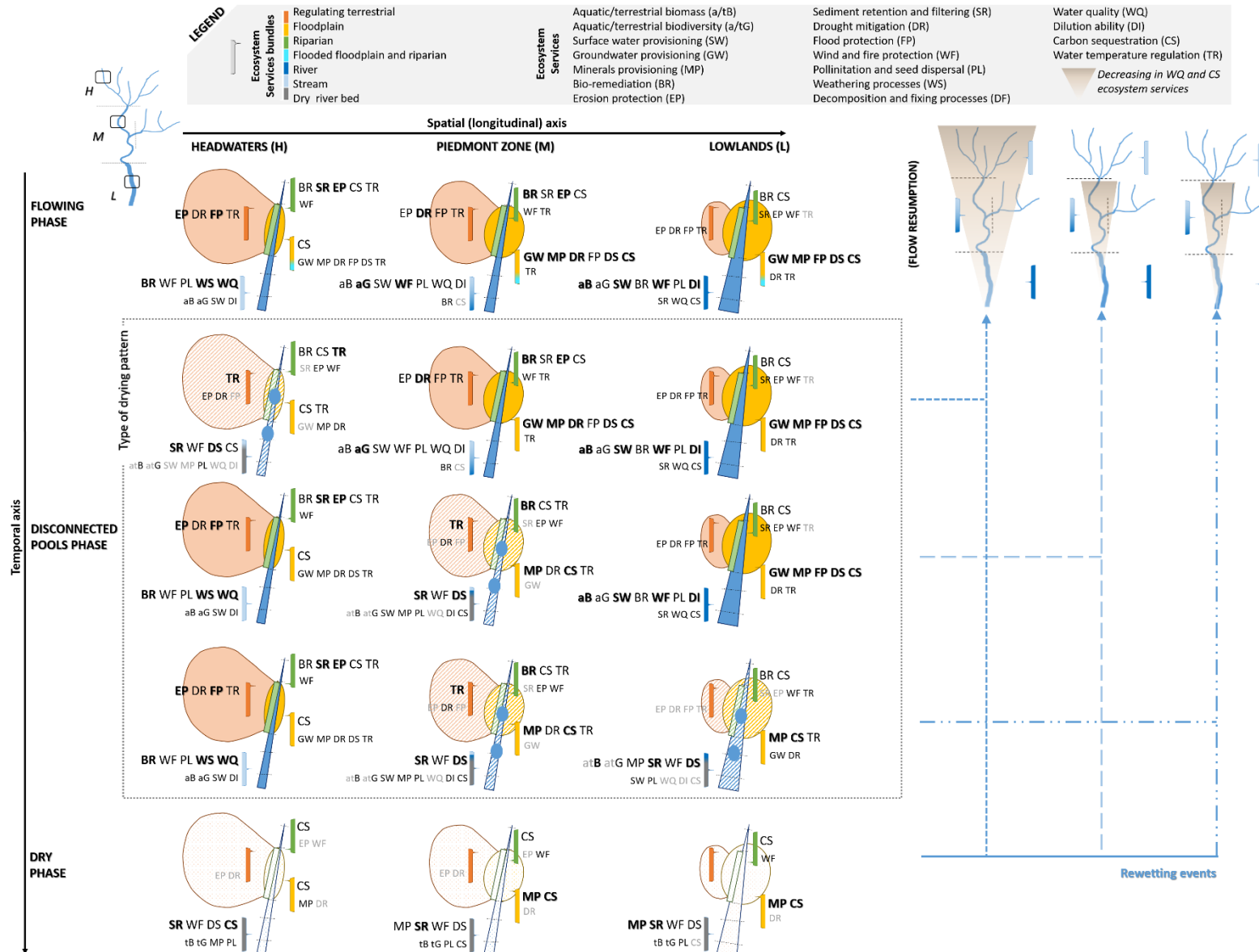
During rewetting events, the hydrological connectivity of the river network can be fully or partially restored depending on the intensity of precipitation (Fig. 16). If the recovery of lateral and hyporheic connectivity is complete, there is also a resumption of river flow and thus recovery of the longitudinal connectivity of the river network (Fig. 15.a). Consequently, the river network transitions back to the flowing phase. While the regulating terrestrial, floodplain and riparian ES bundles are once again very relevant in the terrestrial functional units, river and stream ES bundles dominate the reach functional unit (Fig. 16.a2 and Fig. 17). However, in the case of river and stream ES bundles, the ES provision is conditioned with respect to the flowing phase due to particularities in the meta-ecosystem and meta-community dynamics. On the one hand, flow resumption through the re-establishment of hydrological connectivity also determines the immediate recovery of ES that are mainly driven by abiotic flows (e.g. surface water provisioning, dilution ability or weathering processes; Fig. 14.a). In this sense, the magnitude of the service depends largely on the intensity and duration of the rewetting event. On the other hand, biological communities are in an active phase of recolonization of the newly re-established lotic habitats. During flow resumption, dispersal from refuge areas is very high (e.g. logs, sediments or patches that have remained wet; Chester & Robson, 2011), however the re-establishment of communities is dependent on the dispersal strategies of the species (i.e. drift, benthic migrations or

multidirectional dispersal by flying) and the spatial distribution of refuges supporting source of colonists (Datry et al., 2017). The latter factor depends in turn on the duration of the previous phases, as well as on the drying pattern of the river network. In relation to river and stream ES bundles, the restoration of the biological community conditions the provision of ES narrowly related to biodiversity such as aquatic biomass and biodiversity production, bio-remediation, water quality or carbon sequestration ES (Fig 14.a). Consequently, these ES may not be recover immediately after flow resumption.

Specifically, meta-ecosystem dynamics drive highly water quality and carbon sequestration or emission during flow resumption events. Recovering longitudinal hydrological connectivity involves the downstream mobilization of sediments and materials accumulated during the dry phase, while recovering of lateral connectivity also reactivates CPOM and DOC inputs from drainage wings, floodplains and riparian buffer functional units (Mulholland & Hill, 1997). In the last case, the quantity of transported CPOM and DOC, but also nutrients, from the terrestrial functional units can be highly variable depending on the duration of the previous dry phase and land cover (Kaplan & Newbold, 1993). In this sense, large, sudden pulses of CPOM and DOC transport and storage can produce “hot moments” of biogeochemical transfer and transformation due to re-activation of the allochthonous processing channel (Datry et al., 2014; von Schiller et al., 2017), reducing the quantity of processed material received downstream. However, in many cases the biological communities may not be immediately available to process these high concentrations of mobilized CPOM, which is mostly exported downstream. For example, after prolonged drought events the terrestrialization of macroinvertebrate communities implies a reduction in aquatic fungi and shredder densities, decreasing the decomposition rates (Bogan et al., 2015; Corti et al., 2011). In turn, the processing of the massive input of terrestrial labile compounds and nutrients availability can also cause dissolved oxygen to plummet, killing invertebrates and fishes, emitting CH<sub>4</sub>, and exporting unprocessed material (Hladyz et al., 2011; Raymond et al., 2013; von Schiller et al., 2017). Thus, rewetting and subsequent flow resumption may initially reduce provision of water quality and carbon sequestration ES. These events have a more negative impact when the rewetting of the river occurs rapidly (e.g. after heavy rainfall events) and there is a large amount of accumulated organic matter in the river network. According to our conceptual framework, these events may be more relevant in DRN that have remained dry in the headwaters for a prolonged period of time (Fig. 16.a2 and Fig. 17). On contrary, in the other two drying patterns, the organic matter accumulated in the DRN is lower, as processing has lasted longer in the headwaters that have been kept flowing, and not as much organic matter accumulates in lowlands (Fig. 16.a2 and Fig. 17). In addition, annual balance of CPOM transported to the downstream receiving waters may be lower in these types of drying patterns (Datry et al., 2017).



**Figure 16.** Conceptual model for ES provision in DRN. The model represents the variations in the ES bundles (to identify the ES of each ES bundle, see Figures 14 and 17) for three hydrological phases (i.e. flowing phase, disconnected pools phase and dry phase) along the longitudinal gradient of the river network (i.e. headwaters, piedmont areas and lowlands). Within the flowing phase, the provision of ES is also differentiated in the case of flow resumption after rewetting event (a.2; the brown gradient indicates how the ES provision of the river and stream ES bundles varies depending on the amount of organic matter accumulated in the fluvial network: the higher the amount of organic matter, the lower the provision of water quality and carbon sequestration ES), as well as the effects of a flood event (a.1; using light blue color: while in regulating terrestrial and flooded floodplain and riparian ES bundles blue color shows a greater intensity in the provision of their ES, in river and stream ES bundles shows higher provision of surface water provisioning, lower provision of water quality and bio-remediation ES, and neutral provision of carbon sequestration ES). In addition, the conceptual model contemplates 3 different drying patterns: headwaters (b.1), middle reaches (b.2) and contraction from low reaches (b.3). The conceptual model explains the variations in the ES bundles based on the changes that occur in the dynamics of meta-ecosystems (“flow as vector for connectivity” and “organic carbon”), meta-communities and meta-populations (both characterized by changes in “community assemblage” and “dispersal”) due to river flow variations (“river flow”). The intensity of these changes (e.g. amount of organic matter that accumulates in the network, changes in biotic communities, etc.) depends on the time that elapses between the different phases and events that are contemplated (i.e. type of flow resumption, time between floods when the river network is in moderate flows, time until flow cessation, time in the disconnected pools phase, time in the dry phase and timing-type of the rewetting event).



**Figure 17.** Spatial-temporal variation in the main ES provided and/or delivered by the ES bundles in the different functional units of the catchment. Size and color of the ES represent the intensity with which they are provided/delivered (see Fig. 14 in order to differentiate this information). ES provision/delivery of “regulating terrestrial”, “floodplain”, “riparian” and “flooded floodplain and riparian” ES bundles are comparable between them. ES provision/delivery of “river”, “stream”, “dry river bed” ES bundles are comparable between them.

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## Part 2. Ecosystem services provision under drying conditions and its application at the focal DRN level

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### Abstract

Climate change is one of the most pressing challenges of our time, with serious threats to earth socio-ecosystems, including river networks and their associated catchments. The effects of climate change might emerge as a combination of factors from extended and more intense precipitation or drought events, average temperature increases or extreme heat waves, among many others. These global warming consequences are also interacting with other human pressures, such as deforestation, pollutants and others, generating very serious threats for biodiversity conservation and human societies future development. Thus, implementing effective solutions that can buffer the effects of climate change and help designing adaptation pathways to climate change are much needed.

Ecosystem service modelling emerges as a potential tool to assess the status of regulatory functions in the different landscape spatial units. This framework can be used to detect which areas in the catchment or a river network might need specific restoration or conservation actions to maintain or reach a desired level of ecosystem service provision. The main objective of the second part of this deliverable is to specifically model a selection of six ecosystem services that can be affected by climate change across the six case studies within DRYvER. This second part builds on top of the conceptual model developed under the first part, in which the main ecosystem components and functional units that need to be considered when modelling ecosystem services in DRNs have been described. In more detail, in this part of the deliverable we present a practical methodology for modelling each of the selected ES which is adapted to the type of data available in the project but considering the conceptual model of the first part of the deliverable. Finally, we present the results of the modelling exercise in the 6 case studies of the project and we briefly discuss how the results could be used in the design and implementation of Nature Based Solutions.

### *Keywords*

Ecosystem service modelling, Integrated catchment management, integrated landscape management, terrestrial-aquatic interfaces, river networks, risk management.

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CONTRIBUTORS	¿?

<sup>5</sup> Use one of the following codes:

R=Document, report (excluding the periodic and final reports)  
DEM=Demonstrator, pilot, prototype, plan designs  
DEC=Websites, patents filing, press & media actions, videos, etc.  
OTHER=Software, technical diagram, etc.  
ORDP : Open Research Data Pilot

<sup>6</sup> Use one of the following codes:

PU=Public, fully open, e.g. web  
CO=Confidential, restricted under conditions set out in Model Grant Agreement  
CI=Classified, information as referred to in Commission Decision 2001/844/EC.

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## Introduction

Ecosystem services (ES, hereafter) modelling is a valuable tool used to quantify, map and assess the benefits that humans obtain from ecosystems (Olander et al., 2018). It involves the use of conceptual frameworks, spatial data and methods necessary to understand how ecosystems provide services that contribute to human well-being. The capacity of ecosystems to deliver essential services to society is already under stress because of a wide range of factors that are included in the so-called global change (Mooney et al., 2009). Among all, climate change is one of the most pressing challenges of our time, with far-reaching impacts on various ecosystems, including river basins and river networks. Rising temperatures, changing precipitation patterns, and more frequent extreme weather events are altering the delicate balance of water resources (IPCC, 2014). This changes disrupt natural flow regimes and intensify hydrological variability with significant consequences for natural ecosystems and human societies that rely on them for drinking water, irrigation, hydropower and other socio-economic demands (Zhang et al., 2016). In addition to hydrological shifts, climate change also exerts significant influences on water quality. Increased temperatures, combined with changes in precipitation, lead to altered nutrient cycling, increased sedimentation and changes in pollutant transport dynamics (Dudgeon et al., 2006). These factors contribute to deteriorating water quality, eutrophication and the loss of biodiversity in rivers and associated ecosystems, influencing the availability of suitable habitats, migration patterns of aquatic species and overall ecosystem productivity (Döll et al., 2014). Moreover, climate change-induced shifts in precipitation patterns can result in more prolonged dry spells or intense rainfall events, leading to heightened risks of droughts, floods and associated socio-economic impacts (IPCC, 2018).

River responses to climate change needs an urgent and comprehensive assessment on how altered precipitation patterns and changing temperatures affect these ecosystems. A catchment and river network scale approach is needed for improving our understanding and to be able to design adequate adaptation pathways to climate change impacts (Arthington et al., 2006). This also reinforces the challenge of providing environmental flow rules to sustain river ecosystems in the face of climate change. Altered hydrological patterns and water availability can disrupt the natural flow regimes of rivers, impacting habitat suitability, biodiversity and ecosystem functioning, emphasizing the importance of considering climate change effects when establishing environmental flow requirements for river networks. Vörösmarty et al., (2010) provides an overview of the global threats to human water security and river biodiversity, with a focus on climate change impacts, highlighting how changes in temperature, precipitation and hydrological regimens directly affect water availability, river flow regime and river ecosystem integrity.

The stresses imposed by climate change need also to be considered in a spatial context defined by the current and future configuration of land use and land cover (LULC) patterns at both the catchment and river network scales. The spatial distribution of land cover types and the elimination of mature and stable forest ecosystems by means of historical deforestation, urbanization or agricultural practices, involve modifications in the physical and biological characteristics such as soil composition and water runoff patterns (Neary et al., 2009). This directly impacts river ecosystems through changes in soil erosion and sedimentation, increasing nutrient and pollutant loads, or the quantity and type of organic matter entering rivers (Vörösmarty et al., 2010). The synergetic effects of these LULC changes and climate exacerbate the ecological consequences of climate change, posing a threat to biodiversity, water quality and overall ecosystem health that needs to be recognized as a unique, complex mosaic of interdependencies for designing sustainable management and conservation practices (Lehner et al., 2006). For example, increased rainfall variability may exacerbate the erosion and sedimentation

caused by deforestation, leading to more significant impacts on river ecosystems (Ellison et al., 2017). Similarly, climate change-induced droughts can compound the negative effects of water extraction for agricultural purposes, further reducing water availability and impacting aquatic biodiversity (Reid et al., 2018; Vörösmarty et al., 2010). Understanding how these effects interact is crucial for effective river ecosystem management and conservation. Integrated approaches that account for land use planning, ecosystem restoration and climate change adaptation strategies are essential to maintain the ecological integrity of river systems. To this aim, the part 1 of this deliverable has focused on laying out a conceptual model in which the interconnectivity of the terrestrial and aquatic ecosystems is specifically recognized for different hydrological conditions and in which a number of spatial functional units are laid out for analyzing energy, water and material flows across the landscape.

Adopting integrative approaches and considering future climate scenarios, can help on the design of adaptation pathways that buffer the negative effects of climate change and promote ecosystem resilience. In relation to this, the strategic implementation of Nature Based Solution (NBSs) across landscapes might help addressing the complex challenges the climate change bring. To achieve this goal, ES modeling and assessment across landscapes is a powerful tool that can be used to identify key areas in which regulatory ES need to be enhanced or preserved. ES assessment can be performed at different spatial levels ranging from the entire catchment (such as hydrological regulation, water provisioning and drought regulation), to the areas that connect the terrestrial environment to the river network (e.g. in flooding and erosion regulation) or to the riverine environment itself (e.g. in thermal regulation effects and carbon sequestration). This multiscale analyses can provide valuable insights into the functioning of DRNs and their catchments and the benefits they provide to human societies. In relation to this, a conceptual exercise has already been provided in part 1 of this deliverable to demonstrate how the provision of different groups of ecosystem services might vary across landscape functional units (i.e. spatially) and across hydrological conditions (i.e. temporally).

Moreover, in regions where water scarcity is a concern, the ecological functioning and service supply of DRNs can have serious implications for water resource management. In this regard, modelling ES can help understanding the key hydrological processes and the key water availability issues which can guide decisions on water allocation, water use and water management strategies to ensure sustainable use of water resources while considering the ecological needs of river ecosystems (Datry et al., 2017, 2018; Messenger et al., 2021). This exercise can also help identifying areas that are critical for maintaining key ecological functions and prioritize conservation and restoration efforts. DRNs, where river channels intermittently dry up due to changing hydrological patterns or human activities, are ecologically important as they support unique habitats and species adapted to these dynamic flow regimes. Moreover, modelling ES can help identifying key habitats and areas of ecological importance, which can inform conservation and management strategies to protect these fragile ecosystems (Du et al., 2022; Gounand et al., 2020; Leigh & Sheldon, 2009). DRNs also support high biodiversity, and dryness characteristics can have significant impacts on aquatic species and their habitats. All this information can aid in the development of conservation strategies for protecting vulnerable species and maintaining ecosystem resilience. In relation to Climate Change Adaptation, dry periods are expected to become more frequent and severe due to climate change. Finally, ES characterization can also assist in the development of climate change adaptation strategies, such as identifying areas that are likely to be most affected and implementing measures to buffer climate change impacts.

Thus, in this second part of the deliverable we aim at modelling a diversity of ecosystem services provided by the catchment and the DRN that are supposed to be affected by climate change. The methodological approach proposed in here follows the fundamentals, assumptions and spatial units drawn in the first part of the deliverable, but are adapted to the availability of existing data in the

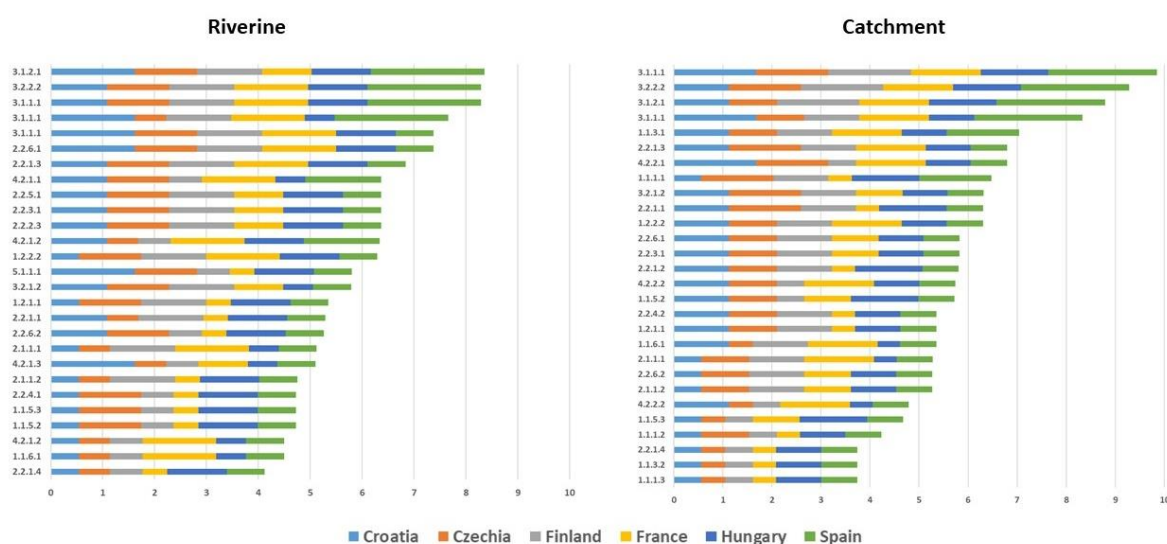
project or in global databases. The selection process involved (described below) six different ecosystem services: Water provision, drought regulation, flood regulation, erosion regulation, thermal regulation and carbon emissions. In relation to water provision this ES refers to the amount of water, in terms of quantity and quality, provided by freshwater ecosystems for consumptive use and for non-consumptive use (MME, 2005). Water provision is now well beyond levels that can sustain current demands (MME, 2005), and given its straightforward dependency on precipitation and evapotranspiration patterns and how water is routed in the catchment, this situation is predicted to worsen under the future climate change scenarios, bringing dramatic consequences for human populations. Secondly, drought regulation is related to the hydrological response of the catchment, linked to the potential to store or hold water and subsequently contribute to water availability over time (De Groot et al., 2002; Haines-Young & Potschin, 2018), which confers resilience to the catchment. In the context of human population growth and increasing extreme climate events, especially rising temperatures and more unpredictable precipitation patterns, drought is becoming more frequent and severe, increasing pressure on water availability and affecting multiple aspects of human life. Thirdly, flooding regulation is the capacity of the catchment ecosystems to reduce and mitigate the impact of extreme precipitation events, i.e. floods, in the downstream parts of the catchment. Flood regulation is becoming increasingly important in the current context of climate change, with the frequency and intensity of floods continuing to rise, because of its relation to the protection of human societies, infrastructure and ecosystem functions. Fourthly, erosion regulation constitutes the role that ecosystems and vegetation play in retaining soil or avoiding soil being eroded as a result of wind or run-off water (Burkhard & Maes, 2017). Accelerated soil erosion is a serious concern worldwide, and it is difficult to assess its economic and environmental impact accurately because of its extent, magnitude, rate and the complex processes associated with it (Erkal & Yildirim, 2012). The impact of natural hillslope processes is important and is currently strongly influenced by human activity due to land use change and vegetation removal and is becoming even greater due to climate change (Van Beek et al., 2008). Fifthly, thermal regulation refers to the potential effect of diminishing the incident solar radiation in the river by the shadow cast by riparian forests. Overall, estimating this ES on a river reach involves combining data on terrain elevation, riparian forest type and location, orientation of the river reach and sun angle and position. Finally, carbon emissions from rivers to the atmosphere depend on in-stream metabolism and groundwater transport of CO<sub>2</sub> from terrestrial ecosystems. Globally, it is estimated to be ~ 1.8 Pt annually and is likely underestimated as non-perennial river networks are not integrated in carbon evasion calculations (Marcé et al., 2019; Raymond et al., 2013). Moreover, change carbon emissions from rivers may increase under climate change conditions due to higher temperatures and longer and more severe drying events. In conclusion, modelling ecosystem services in DRNs and their catchments is crucial for improve our understanding of the ecological, hydrological and socioeconomic implications of climate change.

The main objective of this report is to estimate the provision of six regulatory ecosystem services to inform landscape planning measures in the face of climate change. The modelling of these six ES will constitute a practical framework for understanding the multifaceted nature of ecosystem regulation in the face of climate change. The work developed will allow to provide valuable insights for effective management and conservation strategies at various scales such as entire catchments, river networks and functional units across the landscape (see Part 1 of this deliverable). By incorporating this comprehensive understanding of ecosystem services into landscape planning, participatory approaches or decision-makers can make informed choices and develop sustainable strategies that promote resilience and adaptability. This holistic approach recognizes the interconnectedness of

ecosystems and the crucial role they play in maintaining the health and functionality of river ecosystems. Overall, this idea holds great potential for supporting landscape planning efforts, fostering ecosystem conservation, restoration and effectively addressing climate change challenges.

## Regulatory ecosystem services

The selection of ES to be modelled within DRYvER followed 3 sequential steps. First of all, we generated a full list of potential ES that could be generated by the river network and the whole catchment for each of the six case studies. The list of ES was produced following the codes and nomenclature of the CICES version 5.1 (Haines-Young & Potschin, 2018). Secondly, we handed those lists to the case study leaders so that they could score the importance of the provisioning of those ES within their geographical context. Case study leaders could only cast their votes through a low, medium, high rank. These ranks were later transformed into one, two and three ordinals, and they were normalized for each case study (i.e. not everybody used H or L with an even frequency). Later, the average importance of each ES was calculated by summing up the scores across the six case studies (Figure 1).



**Figure 1.** Average score for the importance of ecosystem service provisioning across the different case studies. The ES that received a higher average score than 6 are the following: Food production (1.1.1.1), Milk or meat production (1.1.3.1), Conservation of the fish genetic pool (1.2.2.2), Conservation of the wild animal genetic pool (1.2.2.2), Soil erosion protection (2.2.1.1), Reduction of flood risks (2.2.1.3), Conservation of nursery habitats (2.2.2.3), Regulation of invasive species or pests (2.2.3.1), Nutrient assimilation (2.2.5.1), Carbon sequestration (2.2.6.1), Recreational activities (3.1.1.1; 4<sup>th</sup> in Riverine and 1<sup>st</sup> in Catchment), Sport fishing (3.1.1.1; 5<sup>th</sup> in Riverine), Tourism (3.1.1.1; 3<sup>rd</sup> in Riverine and 4<sup>th</sup> in Catchment), Education and research on Nature (3.1.2.1), Traditions and myths related to natural locations (3.2.1.2), Personal satisfaction-Bequest value (3.2.2.2), Water for irrigation (4.2.1.2) and Water for human consumption (4.2.2.1).

Finally, in the last step we evaluated which of the above ES could be calculated within the DRYvER context. This meaning that the required datasets were going to be available from the previous Working Packages and tasks, so as to cover a meaningful list of ES related to the DRYvER central issues (e.g. the effects of Climate Change and drying). The final selection included 6 Ecosystem Services encompassing Water provisioning, Hydrological regulation, Erosion regulation, Thermal regulation and Carbon emissions (Table 1).

**Table 1.** Selected ES to be modelled within the project DRYvER. Codes within the Domain heading represent Catchment (C) and Riverine (R) contexts.

Ecosystem Service Name	Type	Domain	Inputs
Water provisioning	Provisioning	C	WP1
Drought regulation	Regulating	C	WP1
Flood regulation	Regulating	R, C	WP1
Erosion regulation	Regulating	R, C	WP4
Thermal regulation	Regulating	R	WP3
Carbon emissions (Ecosystem function)	Regulating	R	WP3

The actual ES models developed across DRYvER DRNs are the following:

1. Water provisioning.
2. Drought regulation
3. Flooding regulation (Hill + Floodplain)
4. Erosion regulation (Hill + Riparian)
5. Thermal regulation (shadow cast by riparian forests)
6. C-Sequestration

We briefly explain below the importance of each of the regulatory ES at different spatial levels, ranging from a catchment level control for 1 and 2, a coupled catchment and river network level (i.e. valley bottoms and riparian forests) for 3 and 4 and specifically following a riparian network structure with river reaches with a different vegetation composition across environmental gradients (5 and 6).

The water provisioning service has received increased attention as water is vital to life on earth (Karabulut et al., 2016) and needed to maintain many critical human activities including water for domestic uses such as washing and cleaning, water for industry and agriculture and water for renewable hydropower generation. It is clear that this ecosystem service is at the core of the water-food-energy nexus (Endo et al., 2017), so mapping and assessing water provisioning in a context of global and social change, including population growth, globalization and economic growth, is paramount to detect potential risk for water security and propose sound water resources management plans. The case of DRN is especially critical, since they are subjected to important spatio-temporal oscillations in the flow regime, i.e. DRNs usually showed a high seasonal climate variability and some parts of the river network get dry while other remain flowing. These issues can cause important contrasts in “when” and “where” this ES can be provisioned. This spatio-temporal variability will increase in a context of global change.

Drought is the result of a complex interaction between climatic, hydrological and socio-economic factors at different temporal and spatial scales, aggravated by climate change and the demand of water for human consumption. The increase in drought events leads to greater pressure on water availability in the basin, which has a negative impact on society and the environment. Water storage mechanisms at the catchment level, whether on the surface, subsurface flows or in groundwater, generate flows in periods of lower precipitation (Mcnamara et al., 2011) contributing to the water availability in the catchment during such periods. Understanding which areas of the landscape are most suitable for storing surface water, or which parts enhance local recharge, is particularly useful to improve the planning and management of the catchment and its water resources. For this purpose, different factors

involved in the hydrological response of the basin and storage mechanisms have been evaluated, such as hydrological factors, land cover and land use, and topography, due to their key role in hydrological regulation. The drought regulation ES developed in this work is a powerful tool for spatially assessing storage mechanisms in the catchment.

Flood regulation refers to the process of mitigation of flood impacts in certain areas of the catchment, especially in the most downstream areas. Flood regulation ES is one of the most important ecosystem services due to its role as protection against extreme meteorological phenomena, particularly in settled areas or infrastructures. Flood regulation ES are intrinsically related to the hydrological response of a catchment and hydrological connectivity, and are dependent on topological, edaphological, geological and biological components as well as the precipitation regime (Nadeau & Rains, 2007). There are mainly two components in the catchment that are of special concern for their capacity to reduce and mitigate the effect of floods, the mature forest on the hillsides and well-preserved floodplains, therefore this ES must be assessed at the catchment control level and at the floodplain level. Flood regulation in slopes refers to the potential contribution of each cell to the occurrence and intensity of peak flows, as well as their control by the presence of forest, in order to prevent flooding downstream. Flood regulation in floodplains refers to the volume of water that could potentially be temporarily stored in the event of a typical flood that overflows the channel. In the current context of climate change, extreme weather events are expected to become more frequent and intense, making the pressure and risk of flooding increase and thus making this ES essential to preserve ecosystem functions and protect society.

Erosion is the process of carrying away or displacement of sediment by the action of wind, water, gravity, or ice (Smith & Smith, 1998). In the case of soil erosion by water, both rainsplash and water running over the soil surface detach and then move the detached particles, but rainsplash is the most important detaching agent whereas running water is the principal transporting agent (Stolte, 2016). Most concerns about erosion are related to accelerated erosion, where the natural rate has been significantly increased by human activity, and the environmental, economic and social impacts derived from it. The effects derived from erosion processes may take place on-site and off-site. The first include the loss of soil, the breakdown of soil structure, the decline in organic matter and nutrient, the reduction of available soil moisture resulting in a decline in soil fertility and in more drought-prone conditions. Off-site problems arise from sedimentation downstream which may reduce the capacity of rivers and drainage ditches, enhances the risk of flooding and shortens the design life of reservoirs. Moreover, sediment is also a pollutant in its own and through the chemicals adsorbed to it resulting in an increase of the levels of nitrogen and phosphorus in water bodies and in eutrophication. In addition, erosion may contribute to climatic change as a consequence of the carbon released into the atmosphere as CO<sub>2</sub> during the breakdown of soil aggregates and clods into their primary particles (Morgan, 2005). For these reasons, water erosion represents one of the most important and widespread causes of soil degradation in Europe (Alcamo et al., 2007). The main factors that contribute to soil erosion are rainfall, erodibility or soil type, absence of vegetation, slope and land management. In particular, Mediterranean type ecosystems are heavily affected by intense soil erosion processes due to high rainfall intensities, recurrent droughts, erodible parent materials and a long history of abuse of soil, water, and vegetation resources (Keesstra et al., 2018). Erosion may also be exacerbated in the future in many parts of the world because of climatic change towards a more vigorous hydrologic cycle (Amore et al., 2004; Pandey et al., 2007). Erosion regulation ES makes reference to the ability of the ecosystems to prevent and mitigate soil erosion (Burkhard et al., 2014). In this sense, vegetation

can act as a protective barrier between the soil and the natural elements which stimulate erosion or mass movement by improving slope stability through changes in mechanical and hydrological properties of the root-soil matrix (Stokes et al., 2008). In particular, forests play the role of Service Providing Areas (SPA) as trees contribute importantly to control soil erosion by reducing the erosivity power of precipitation (Burkhard & Maes, 2017). Riparian zone is very important in river management, because the vegetation in riparian zones carries out important ecological, hydraulic and hydrological functions, such as sediment trapping to prevent it from entering streams (Sheridan et al., 1999).

Thermal regulation, enabled by the shadow cast by riparian forests, is a crucial factor in maintaining the health and functioning of aquatic ecosystems (Broadmeadow & Nisbet, 2004). The dense foliage of riparian trees acts as a natural barrier, casting shadows over the water surface and providing relief from direct sunlight exposure. This shading effect plays a vital role in mitigating temperature fluctuations in valleys and streams. By reducing the amount of solar radiation reaching the water, riparian forests help to regulate water temperature and prevent overheating during periods of intense sunlight and high air temperatures, especially during hotter summer months, which can prevent harmful algal blooms, fish die-offs and the survival and reproduction of many aquatic organisms (Bachiller-Jareno et al., 2019). Shade is also important to create ideal habitat conditions for many aquatic organisms by helping promoting the growth of aquatic plants, which provide cover and food for fish and other aquatic organisms (Bonacina et al., 2023). Riparian forests also contribute to cycle nutrients between the land and water. As a result, less nutrients are needed to support the growth of algae and other photosynthetic organisms, which can help prevent eutrophication and maintain water quality. Overall, the shadow cast by riparian forests plays a critical role in maintaining the health and ecological integrity of river ecosystems, supporting the diverse array of aquatic organisms that depend on healthy rivers for survival.

Rivers are essential to the global carbon cycle by producing, processing and transporting organic and inorganic carbon. Organic carbon storage is found in floodplain sediments, riparian vegetation, dead wood and in-stream biomass. The only strictly fluvial sink of carbon is in-stream biomass, which is calculated to be the least important due to the high dynamic nature of river ecosystems (Sutfin et al., 2016). While sediments, riparian vegetation and floodplains are considered carbon sinks, the river channels are generally heterotrophic and net emitters of inorganic carbon into the atmosphere (Battin et al., 2023). The inorganic carbon present in rivers has two main origins: 1) internal production due to the decomposition of organic matter by heterotrophic organisms and 2) external CO<sub>2</sub> originated in the surrounding terrestrial ecosystems and transported through groundwater to the river. A river emits inorganic carbon into the atmosphere if the sum of external and internally produced CO<sub>2</sub> is higher than the CO<sub>2</sub> fixated into organic carbon through photosynthesis (Hotchkiss et al., 2015). Until now, carbon emission studies have focused on perennial streams and rivers, not considering the effect of intermittent and ephemeral water courses (Raymond et al., 2013). Recent studies suggest this omission may be underestimating the CO<sub>2</sub> evasion from rivers into the atmosphere as dry sediments are more heterotrophic than flowing waters (Marcé et al., 2019).

## Methodology for modelling ES provision

In this section we describe the different characteristics of the geographical context from each case study and the methodology performed to estimate the provision of the 6 selected ecosystem services.

## Case study description

The selected ES developed in this report are spatialized across Europe in 6 DRNs representing different biogeographic and climatic settings at two different scales depending on the ES, whole catchment/focal DRNs catchments, referred to as:

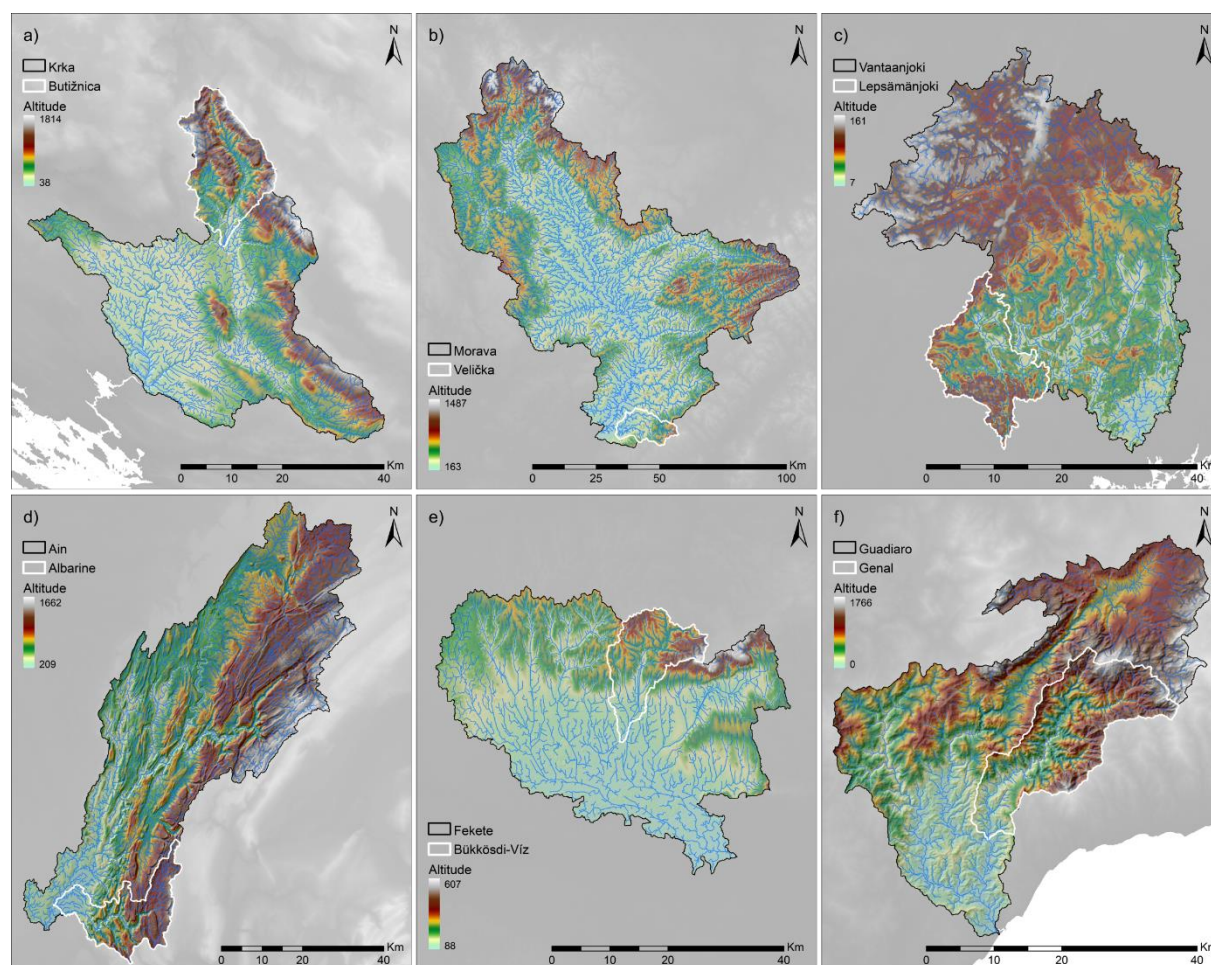
- a. Krka/Butižnica (Croatia; CR). Mediterranean climate. Dry region heavily impacted by climate change, where most rivers are already affected by drying.
- b. Morava/Velička (Czech Republic; CZ). Continental climate. Region heavily impacted by climate change where many perennial rivers are shifting towards intermittent.
- c. Vantaanjoki/Lepsämäenjoki (Finland; FN). Boreal climate. Region moderately impacted by climate change, where flow intermittence is currently rare.
- d. Ain/Albarine (France; FR). Temperate climate. Region mildly impacted by climate change
- e. Fekete/Bükkösdi (Hungary; HU). Continental climate. Region moderately impacted by climate change, where DRNs are becoming common.
- f. Guadiaro/Genal (Spain; SP). Mediterranean climate. Dry region heavily impacted by climate change, where most rivers are already affected by drying.

The flood regulation, erosion regulation, drought mitigation and thermal regulation ES were performed in the large river basins of the six countries (Krka, Morava, Vantaanjoki, Ain, Fekete and Guadiaro), while water provisioning and carbon emissions ES were performed in the surveyed DRNs which are a smaller portion of the larger basins (Butižnica, Velička, Lepsämäenjoki, Albarine, Bükkösdi and Genal).

The assessment of climate change effects and more specifically summer drought on river function and services is of paramount importance in the management of these selected rivers and associated ecosystems for several reasons that share all case studies. First, we need understanding climate change impacts on ecosystem function. Climate change has the potential to significantly alter precipitation patterns and increase the frequency and intensity of drought events. A comprehensive understanding of the potential impacts on rivers and surrounding ecosystems is crucial for developing appropriate management strategies that take into account the changing hydrological conditions. At the level of water resource management, all selected rivers serve as a water resource for various purposes, including drinking water supply, irrigation or industrial purposes. Drought events associated with climate change can reduce resource availability and intermittence, jeopardizing further uses and ecosystem functionality. By assessing these climate change and drought effects, water managers can anticipate potential water scarcity issues and droughts and implement proactive measures to ensure sustainable water management practices during river intermittence episodes. Rivers and related ecosystems support across case studies a wide array of plant and animal species. Climate change and drought can negatively impact the entire ecosystem by altering water flows, reducing water quality and disrupting ecological processes. Assessing their effects will allow us to identify vulnerable areas and develop conservation strategies to protect the ecological integrity of the rivers and its associated habitats. There also exist socioeconomic considerations, since selected rivers may play a significant role in supporting local economies through tourism, recreational activities and fisheries. Climate change and drought can disrupt these socioeconomic activities by altering the river's characteristics and availability of water resources, so there is a need for understanding the potential socio-economic impacts that may affect the development of adaptation measures to minimize negative consequences on local communities and economies. Finally, at the level of planning and adaptation, an effective management of the river basins and riparian networks selected may require long-term planning and adaptation strategies, so we need essential information of climate change-related drought effects for

incorporating climate resilience into management plans. By understanding future climatic scenarios and their implications, decision-makers can implement adaptive measures such as water conservation practices, water demand management and ecosystem restoration initiatives.

In this context, the assessment of regulatory ecosystem services has been conducted at different spatial levels on which they may affect ecosystem functionality and on which we can apply adaptive solutions based on nature and intelligent spatial planning. Those ES related to hydrological flows (i.e. drought and flooding regulation), erosion regulation and thermal regulation have been evaluated for the whole catchment. Water provisioning and carbon sequestration ES have been evaluated for the focal DRNs. Figure 2 shows the whole and focal catchments, altitudes and river networks for all case studies of the DRYvER project. Table 2 shows a summary of the main factors involved in the assessment of ecosystem services.



**Figure 2.** Overview of case studies included in the ES assessment for the DRYvER project showing the complete catchments (black), focal DRN catchments (white), altitudinal regime and river networks for: a) Croatia, b) Czech Republic, c) Finland, d) France, e) Hungary and f) Spain.

**Table 2.** Summary of key factors of the DRYvER project's ES assessment. The Range for altitude, precipitation and temperature is calculated across tributaries.

	CROATIA	CZECH REPUBLIC	FINLAND	FRANCE	HUNGARY	SPAIN
<b>ALTITUDE (m)</b>						
<b>RANGE</b>	38 – 1814	163 – 1487	7 – 161	209 – 1662	88 – 607	0 – 1766
<b>MEAN</b>	478	404	77	671	154	551
<b>AREA (km<sup>2</sup>)</b>						
<b>CATCHMENT (FOCAL DRN)</b>	1915 (325)	9435 (214)	1666 (208)	3647 (350)	1784 (190)	1480 (336)
<b>RIVER NETWORK LENGTH (km)</b>						
	2739.03	8695.18	2327.01	4070.26	2406.37	1501.20
<b>ANNUAL PRECIPITATION (mm)</b>						
<b>RANGE</b>	759 – 1264	533 – 1090	622 – 688	959 – 1576	605 – 715	677 – 889
<b>MEAN</b>	937	652	659	1219	659	752
<b>ANNUAL MEAN TEMPERATURE (°)</b>						
<b>RANGE</b>	4.8 – 14.9	1.7 – 9.45	3.8 – 5.2	4.5 – 11.4	9.1 – 11.4	10.4 – 17.8
<b>LAND USE AND LAND COVER (LULC) (%)</b>						
<b>URBAN</b>	1.07	7.49	17.21	3.25	5.25	8.78
<b>BARE</b>	3.76	-	-	0.13	-	10.61
<b>AGRICULTURE</b>	19.91	47.44	27.85	10.25	56.55	9.24
<b>GRASSLAND</b>	25.66	8.55	0.04	21.38	6.15	9.02
<b>SHRUBLAND</b>	6.30	0.01	-	0.05	-	15.29
<b>WATER</b>	0.51	0.3	2.13	1.27	1.07	-
<b>CONIFEROUS</b>	1.70	15.08	13.84	17.42	0.29	4.68
<b>FOREST</b>	41.10	21.13	38.93	46.25	30.68	42.39

## Ecosystem service characterisation

Below we explain in detail the methods and datasets used for estimating the provision of each of the six selected ES. We also comment on the limitations and findings of each approach.

### Water provisioning

The ES water provisioning is directly related to the hydrological response of the catchment, as a result of the movement of water through the catchment from a precipitation event to the river network. In DRYvER, the ES water provisioning refers to the amount of superficial water circulating (i.e. available) at a given time in the river network. This means that, other water sources, as groundwater, or water stored in lentic aquatic ecosystems, e.g., lakes, ponds or reservoirs, have not been considered. The ES water provisioning is directly related to the hydrological response of the catchment, as a result of the movement of water through the catchment from a precipitation event (Park et al., 2011) to the river network.

### Background hydrological information

The ES water provisioning is based in the analysis of the simulated discharge generated for each reach of the six DRN in WP1 for the period 2005-2022 (Künne et al., 2022). In summary, WP1 developed a hybrid hydrological modeling approach to simulate daily discharge in the six DRNs. This approach is based on the JAMS/J2000 modeling system/hydrological model, running at a daily time-step. As a first

step, WP1 collected, analyzed and selected the required data to set-up the physical-based model. In a second step, these models were calibrated and validated using long-term (at least, 20 years) streamflow data. ) streamflow data (Künne et al., 2022).

Regarding the first step, WP1 used two types of data: 1) dynamic hydro-climatic time-series of the last 20 years and 2) static spatial topographical, pedo-lithological and land use land cover (LULC) data. The first group of variables were based on the reanalysis of ERA5-land series to provide daily series of precipitation, air temperature and reference evapotranspiration. Additionally, daily discharge data recorded at gauge stations for the six DRNs were collected to calibrate and validate the hydrological model. The second group of variables was used to delineate the modeling entities and deriving bio-physical information for the mathematical calculation of hydrological processes. For the delineation of modeling entities, the concept of Hydrological Response Units (HRUs) and stream segments (reaches) as well as topological routing was used. The HRUs and the synthetic river networks delineated were constrained by the DEMs used for each DRN and validated using the reference river networks provided by the DRN leaders. Besides, bio-physical information was used to calculate plant-based processes (e.g. evapotranspiration) and soil water processes (e.g. infiltration and groundwater processes). This information included topographical variables (extracted from the DEMs), soil classes and parameters (adapted from the European Soil Database v2.0), LULC (based on Corine Land Cover version 2020\_20u1 Level 3 to establish LULC classes and different extra sources to adapt associated parameters) and hydrogeology (based on International Hydrogeological Map of Europe and different literature references to parametrize the groundwater component in the models).

Before moving to the step 2 (model calibration and validation), WP1 defined the model structure. In this regard, the models for the six DRNs were developed using the process-oriented model J2000 together with the modular object-oriented modeling system JAMS accounting for surface, sub-surface, and groundwater flow from hillslopes into the stream and along stream segments until the outlet. Modifications from the standard J2000 hydrological model were made for DRYvER using a specific evapotranspiration and snow modules. Additionally, adaptations were made in the French DRN to account for the influence of the Vouglans hydroelectric dam.

Finally, several techniques were used to calibrate the hydrological models of the six DRNs to find the optimal models and gain a wider perspective of the model's sensitivity, parameter interactions, and uncertainty by investigating hydrological process patterns. These techniques included the calibration of snow parameters and discharge, including both calibration of snow parameters and discharge.

After these processes WP1 provided discharges in  $\text{m}^3/\text{s}$  at daily time step for the period 2005 to 2022 for all the DRN reaches. According to the WP1, the multi-gauge, multi-objective validation of the six models showed satisfactory to good model performances for all DRNs. Finally, it should be remarked that future hydrological datasets incorporating different climate change scenarios could be easily added to this characterization approach.

### Spatio-temporal analysis of ES water provisioning

In order to report the ES water provisioning, we conducted an analysis of the water availability ( $\text{hm}^3$ ) in each DRN by selecting a representative number of river reaches, both perennial and non-perennial, grouped by their contributing catchment area. In addition, we considered two contrasting seasons (wet and dry) to report water availability. In summary, the ES water provisioning analysis followed the following procedure:

1. **Reach selection:** Given that the contributing area to a river reach is a critical factor determining the river reach discharge we segregated each DRN in three types of reaches:

headwater, middle and lowland reaches. This segregation was based in the distribution of frequencies of the river reaches according to their catchment area.

In order to represent the three types of reaches while avoiding introducing a large variability in the analyses we defined headwater, middle and lowland reaches as those corresponding to the percentiles 25, 75 and 90, respectively. Once these percentiles were calculated we selected all the river reaches within  $\pm 20\%$  of these values to select a representative number of reaches in each size category for further analyses.

2. **Reach correspondence:** We defined the correspondence between the river reaches of the fluvial network created in WP1 (defined above) and the synthetic river network created with NetMap under WP4. Networks developed in WP1 and WP4 did not have the same river reaches, preventing a 1:1 assignment. To solve this problem, we followed several strategies to generate the best relationships between identifiers in the two networks. The basic criteria for the assignments have been the distance from one network to another. However, there were 1) areas where networks were similar but the automatic assignment process did not work correctly and ii) areas where the networks showed large discrepancies. In those cases, the automatic transfer was manually checked and corrected when needed.
3. **Segregation of reaches according to flow intermittency:** Based on the flow series transferred to the selected reaches, we segregated them based on their perennial (PR) or non-perennial (NPR) character. PR river reaches were defined as those with more than 95% of the days in the discharge daily record (over 17 years) with flowing water while NPR were defined as those with less than 75% of the days with flowing water. These thresholds were defined in order to observe the differences in water provisioning between PR and NPR.
4. **Quantification and assessment of water provisioning:** For the six groups of selected reaches (3 catchment size classes x 2 types of perennial character), we calculated the total water resources ( $\text{hm}^3$ ) for each year of the flow series (sum of daily flow estimates in a year). In addition, aiming to observe potential intra-annual variability of this ES, we calculated the total water availability for two contrasting hydrological seasons: wet season (January-March) and dry season (July to September).

## Flood regulation

The precipitation regime is one of the most important factors determining the amount of water inflowing in a catchment. However, hydrological responses depend not only on precipitation but also on other hydrological processes that are controlled by topological, geological, edaphological and biological characteristics (Nadeau & Rains, 2007). These components determine the water balance in a given catchment and the water paths into the river network. They are intrinsically linked to runoff, infiltration, percolation and evapotranspiration processes and, therefore, to the probability of flooding after an event of intense precipitation. In low regulation-capacity scenarios (e.g. deforested landscapes), heavy rainfall generates a higher percentage of quick flow, making the rainfall-to-flow transfer period very short and increasing the probability of flood events. In contrast, when the landscape incorporates storage compartments (e.g. aquifers, forests, wetlands, floodplains, etc..) a higher fraction of the precipitation is transformed on slow flow that might reach the river network well after the precipitation event is over (Mcnamara et al., 2011). In this study, the provision of the flooding regulation ecosystem service has been linked to two main components of the landscape: natural mature forest on hillslopes and well conserved floodplains.

Forests are key habitats for maintaining key ecosystem functions relevant to aquatic ecosystems, such as modifying soil structure, allowing better infiltration and water retention, maintaining more stable runoff levels during the drying season (Watson et al., 2018), playing a decisive role in the overall water cycle by balancing infiltration–evapotranspiration–runoff and reducing soil erosion and sediment transport towards receiving streams (Neary et al., 2009). Specifically, some studies have shown how catchments with more than 30% of mature forest have higher hydrological stability, i.e. lower peak flows and higher base flows (Belmar et al., 2018). Therefore, the presence of mature forests on drainage slopes is considered a relevant factor in the provision of ES related to flood regulation. In relation to the second component, the lateral connection between the river and its floodplain naturally reduces flooding in downstream areas by laminating peak flows, allowing overflow and, thus, smoothing the flood wave. Again, vegetation, especially floodplain forests, also play an important role in storing water during floods by increasing the roughness of the floodplain, which in turn increases the residence time of water on the floodplain (Dadson et al., 2017).

For the assessment of the flood regulation ES, we divided the study areas into a set of functional units (see first part of this report). In this case, we used drainage wings, which are adjacent slopes draining into each river segment individually, and *valley bottom* areas (i.e. floodplains; Benda et al., 2007). The use of these functional units allows transferring digital information to the relevant spatial units in which the ES provision is generated.

### Flooding regulation in slopes

We assessed the flood regulation ES through the potential capacity of a hillside to generate surface runoff, as the path of water through the different biophysical components of the catchment determines the spatio-temporal pattern of floods and drought episodes. The potential of a hillside to generate runoff has been considered based on the abiotic factors that primarily control infiltration at the hillside scale (Hopp & McDonnell, 2009): slope and soil permeability. In addition, according to Maetens et al., (2012) areas with high annual rainfall tend to have a more even distribution of rainfall throughout the year, leading to seasonal soil saturation and a higher probability of producing runoff (Ponce & Hawkins, 1996). With these components, we constructed a Hydrological Regulation Index (HRI) that considers the contribution of each cell to the frequency and intensity of peak flows, as well as their potential mitigation by vegetation. On the one hand, the susceptibility to runoff generation was quantified by considering the process of oversaturation of the soil. For this purpose, we multiplied the slope by the inverse of the saturated water content of the first soil layer and we normalized it (factor 1). On the other hand, we incorporated as a second factor (also normalized from 0 to 1) the formulation of the lateral flow travel time (Terink et al., 2015). According to this formulation, the lateral flow travel time depends on the field capacity, the water content when the field is saturated, as well as its conductivity at saturation. As a longer lateral flow travel time will result in a smoother streamflow hydrograph, we incorporated its inverse value in the index. The HydroSoil v2.0 dataset (Simons et al., 2020) was used as the source of the data related to soil hydraulic properties used in the spatial modelling of this ecosystem service, i.e. saturated water content, field capacity, and conductivity at saturation, at aggregated topsoil level (0-30 cm). Furthermore, as noted above, we also incorporated the spatial distribution of precipitation as a third factor using a normalized mean annual precipitation grid. Annual precipitation data were obtained from ERA5Land. Factors 1, 2, and 3 were combined to generate an index of the potential hydrological response of the catchment. The HRI, ranging from 0 to 1, was obtained for each drainage wing in each case study. A high HRI evidences low regulation capacity, i.e. precipitation is converted into surface runoff quickly and directly, increasing the risk of flooding, and decreasing infiltration rates and water supply in dry periods (Martínez-Retureta et al., 2020). In contrast, lower HRI values represent areas where less surface runoff is generated and the risk of flooding is lower.

Finally, we incorporated the capacity of the native forest to regulate flows (i.e., the "sponge effect"; Belmar et al., 2018; Peña-Arancibia et al., 2019), improving the infiltration capacity of surface soils (Bruijnzeel, 2004; Ulrik Ilstedt et al., 2007) and the water retention (El Kateb et al., 2013), and reducing runoff and slowing down the hydrological response of the watershed. Data from CORINE LAND COVER 2018 were used to extract the natural forest cover pixels. We reclassified the potential hydrological response of the previously obtained index from these land use and land cover data using the following rule: if a pixel was covered by natural forest we set the positive value for the index, otherwise we multiplied the index by -1. Therefore, positive values will be related to the presence of forest and are proposed to be conserved, as they represent areas where the flood regulation ES is already being supplied, while negative values (no forest) are proposed as areas to be restored (i.e. potential areas to generate the service). The results were aggregated at the drainage wing level by averaging the pixel values. Finally, HRI scores were classified to evaluate the importance of both conservation and/or restoration actions. The classification was made using the percentiles 25, 50 and 75, where the top class has the highest priority.

### **Water storage in floodplains**

The regulation of the hydrological response in floodplains was quantified in terms of water storage capacity, i.e., the volume of water that could potentially be temporarily stored in the event of a typical flood that overflows the channel (natural connectivity of the floodplain with its respective river channel). The delineation of the floodplain and the potential volume of water stored during a flood event were estimated with the "Valley Floor" and "Flood Storage" tools belonging to the NetMap software (Benda et al., 2007). In this sense, floodplains were delineated using a geomorphological criterion based on the valley surface at a height of "n" times the bankfull depth of each river channel (Fernández et al., 2012). For this project, a multiplicative value of 5 was used as it is related to a flood of approximately 300-year return period (Ilhardt et al., 2000). The value of bankfull channel depth was estimated for each segment of the river network by means of a regional regression between drainage area, mean annual precipitation and field measurements of bankfull channel depth (for more detailed information, see Benda et al., (2011)).

The volume of stored water is defined from a reference plane below which water remains temporarily stored. The reference plane was considered as the average altitude that the water would reach when the entire floodplain polygon is inundated. In this case, we defined the water level as the average elevation above the channel of all pixels contained in the floodplain polygon corresponding to five measurements of bankfull channel depth. Thus, for each floodplain functional unit, the Flood Storage tool calculated the volume of water stored between the height of the reference plane plus its mean elevation above the channel segment and the mean altitude of the channel segment. Once the volume of each floodplain polygon was calculated, they were grouped into larger and hydrologically independent units following the criteria described in Benda et al., (2011).

### **Erosion regulation**

We contemplate the mitigation of the erosion processes by means of two interrelated processes: a) the erosion regulation at source prevents soil loss and avoids the generation of sediment that can be transported (erosion regulation at source in slopes), and b) the erosion filtering process, which retains part of the particles once the transport flow has started and reduces the amount of sediment that is finally delivered to the river network (erosion transport and filtering). In both processes, the effects of the forest as a provisioning element of the erosion regulation ES were considered differently. On one hand, the presence of forests on hillsides was analysed for the reduction of the amount of sediment generated by intercepting rainfall and reducing the velocity of runoff. On the other hand, the cover of

riparian forests was used to quantify the reduction of the amount of sediment delivered to the river network.

### **Erosion regulation in slopes**

Erosion in the form of landslides, gully formation, and surface erosion is driven by slope and topographic convergence (Sidle et al., 1985). Thus, the erosive potential in a given catchment and the probability of sediments reaching the fluvial network and other bodies of water were estimated by applying a dimensionless topographic index on a 5 m resolution DTM. This index uses slope and topographic convergence following the relationship proposed by Miller & Burnett (2007):

$$GEP = (AI * S) / b_i$$

where GEP is the generic erosion potential (from 0 to 1),  $b_i$  is a measure of local topographic convergence (the length of an elevation contour traversed by the flow of a pixel  $i$ , where lower values imply a converging slope), AI is a measure of the local contributing area (within a pixel length) and  $S$  is the local slope. Higher values of the index indicate a greater erosive potential. To calculate the potential sediment input to the river channels and wetlands, the downward flow path from each pixel to the first watercourse or water body was plotted. Using the “Delivery” tool of NetMap software, this process further determines the probability of sediment transfer to the water body associated with its drainage wing. The calculated probability that the sediment is transferred to each downstream pixel decreases monotonically as a function of the gradient and topographic convergence. In this sense, the probability (from 0 to 1) that the sediment reaches its respective riparian zone functional unit was assigned to the source pixel as a multiplicative factor of the pixel's GEP. Thus, not only the erosive potential of the drainage wing was estimated, but also the probability that the generated sediment could potentially be transferred to a river reach.

The effects of the forest were incorporated as a provisioning element of the erosion regulation ES at source, as they contribute to the reduction of the amount of sediment generated. When a pixel is covered by a natural forest class, we considered that erosion is much less than in pixels without this type of vegetation. In this case, the erosion probability (i.e. the GEP) that is avoided at the source by the presence of tree cover in the pixel was quantified. Furthermore, these pixels do not accumulate towards the river reach. Thus, the cumulative delivery for forested pixels in each wing informs about the potential erosion reduction by the forest. On the contrary, the cumulative delivery for the non-forested pixels in each drainage wing informs about the erosion potential due to the lack of forest. Prioritization of the functional units was performed by using the 25th, 50th and 75th percentiles of the cumulative delivery for forested and non-forested pixels independently, resulting in four categories of drainage wings proposed for conservation and four categories of wings proposed for restoration respectively.

### **Erosion regulation in riparian areas**

The erosion regulation ES in riparian zones was considered as a filtering capacity of sediments coming from adjacent hillslopes performed by riparian forests. To estimate this capacity, we previously calculated the delivery from drainage wings with absence of hillside forest (see above). Thereby, only delivery values from pixels whose erosion was not reduced by the presence of forest were considered or, in other words, GEP value potentially delivered to the river reach from the drainage wing lacking natural forests. This allowed identifying river reaches with potentially higher or lower amount of sediment reaching the river network.

Firstly, in order to identify the riparian forest, it was applied a GIS-based geomorphologic approach. According to this, it was established a flood-prone area at 1.25-bankfull depth over the digital elevation

model so that all forest area within those limits was considered riparian forest. A factor multiplying bankfull depth of 1.25 was selected as it was the value that best matched the 50-year flood in deep v-shaped valleys (Fernández et al., 2012). Secondly, the cover of riparian forest inside each drainage wing was quantified. It was assumed that those drainage wings dominated by forest (cover  $\geq 60\%$ ) would contribute effectively to the provision of the erosion regulation ES in riparian areas (i.e. conservation of natural features), while the rest of drainage wings (forest cover  $< 60\%$ ) were considered as failing to provide a satisfactory level of this ES (i.e. restoration of natural features). Finally, both groups (conservation and restoration) were prioritized based on the 25th, 50th and 75th percentiles of the GEP value potentially delivered to the reach from the drainage wing.

### Drought mitigation

The drought regulation ES was considered as a balance between the water inputs and outputs in a catchment and the surface and groundwater storage potential. The amount of water inflow is largely influenced by the climate pattern, specifically precipitation, while the amount of water outflow is a function of the loss by direct surface runoff (known as quick flow) and evapotranspiration. In this sense, the estimation of drought regulation potential in a given catchment could be related to the identification of landscape areas where (1) surface water could potentially be stored (e.g. in wetlands, lakes or ponds) or where (2) infiltration is maximized, so that slow flow and groundwater recharge processes are favored.

### Surface Storage Index

The potential of a catchment to store surface water is related to certain characteristics that allow retaining water in times of precipitation and releasing it later when drought events unfold. Vegetation cover, soil type, and topography are the main factors related to this functionality (De Groot et al., 2002). The proposed method consisted in building an index that evaluates the suitability of the terrain for surface water retention and storage. This index considers the topographic wetness index (TWI), the slope and plan curvature, the land use and land cover and the intensity of precipitation.

Slope, plan curvature and TWI are topographic variables and they were all calculated from 25 m spatial resolution DEM (EU-DEM: Digital Elevation Model over Europe - <https://land.copernicus.eu/>). TWI is widely used as a proxy to identify areas that accumulate soil moisture (Kopecký et al., 2021). This index is derived from a DEM according to the equation proposed by Beven & Kirkby, (1979):

$$TWI = \ln \frac{a}{\tan Slope}$$

where  $a$  is the upslope contribution area that drains into a specific cell, calculated using a flow accumulation algorithm per unit contour length and the local slope defined as the slope between the focal cell and a cell further downslope (i.e.  $\tan Slope$ ; Gruber & Peckham, (2009). Therefore, TWI is an approximation of upslope water supply and downslope water drainage (Beven & Kirkby, 1979), where higher values would mean higher relative soil moisture. In relation to slope, we divided this variable in two categories: slopes less and greater than  $15^\circ$ , since slopes greater than  $9^\circ$  ( $\approx 15^\circ$ ) have been related to lower flow residence times and higher probability of quick flow. Following, plan curvature is a variable related to the behavior of surface flow and water content in the soil and it relates to the curvature of the ground in a vertical plane (Burt & Butcher, 1985). Negative values represent surfaces where the flow converges thus generating accumulation.

Land use and land cover also influence the soil moisture conditions and water availability in the catchment. The forest has a flow-regulating behavior since it acts as a "sponge" that absorbs water during rainy periods and releases it during dry periods (Belmar et al., 2018; Bruijnzeel, 2004; Peña-

Arancibia et al., 2019). This contributes to water retention and the maintenance of base flows. For the characterization of this ES we used CORINE LAND COVER 2018 data. Finally, precipitation is considered as the indicator of the potential magnitude of storage and it was added as mean annual precipitation.

The storage capacity index has been constructed for each DRNs, as a combination of the variables considered. First, each variable was transformed to a common scale from 1 to 10, based in its suitability holding surface water, where values of the variable that are most preferred were assigned to higher suitability values. TWI and precipitation variables were transformed using a linear function, scores from 1 to 10 were progressively assigned as the values of the variables increased. Slope was reclassified in 2 classes, where slopes less than  $15^\circ$  were assigned to a suitability of 10 and slopes greater than  $15^\circ$  had a value of 1. Plan curvature was also reclassified in 2, negative values were rated as 10 and positive values as 5. LULC data was transformed using their unique categories, i.e. forest and water patches had a suitability value of 10 and urban land use a value of 1. Once all the variables were in a common scale they were combined, using the same weight for all the variables. Finally, the index generated was normalized to obtain it ranging from 0 to 1, where higher values of the index represent areas more suitable for hosting surface water. The results have been transferred to each drainage wing to identify the landscape areas where solutions related to water retention and storage could potentially be implemented (e.g. generation of ponds, wetlands, etc.). Drainage wings have been prioritized based on their potential to store surface water, using the percentiles 50 and 75.

### Local Recharge Index

The drought regulation function has also been approximated through the calculation of a local recharge index that allows identifying landscape areas that can contribute to higher groundwater recharge. Local recharge is defined as the part of precipitation that infiltrates the soil and contributes to subsurface and underground water flows. Both types of flow are related to the hydrological response of the catchment. Moreover, the underground water flow or base flow are the most important contribution to river surface flows during the dry season. In this work, local recharge has been calculated using the Seasonal Water Yield model that is part of the inVEST (Integrated Valuation of Ecosystem Services and Tradeoffs; Sharp et al., (2019) software. The data required to run the model are a DEM, monthly precipitation and evapotranspiration maps, LULC maps, hydrological soil groups, area of interest (basin), biophysical table with crop coefficients (Kc) and curve numbers (CN) for each class of LULC and hydrological soil group and precipitation events tables (table 3).

**Table 3.** InVEST model inputs for all the case studies.

Model inputs	Source
DEM	25 m spatial resolution EU-DEM ( <a href="https://land.copernicus.eu/">https://land.copernicus.eu/</a> )
Precipitation	WorldClim (Fick & Hijmans, 2017, available in <a href="https://www.worldclim.org/">https://www.worldclim.org/</a> )
ET <sub>0</sub>	Calculated according to Hargreaves equation (Hargreaves & Samani, 1985) from WorldClim data (Fick & Hijmans, 2017) <a href="https://doi.org/10.1002/joc.5086">https://doi.org/10.1002/joc.5086</a>
LULC	CORINE LAND COVER 2018 ( <a href="https://land.copernicus.eu/">https://land.copernicus.eu/</a> )
Soil hydrological groups	HiHydrosoils (Simons et al., 2020)
Biophysical table	Kc values: FAO guidelines (Allen et al., 1998) CN values: USDA Handbook Chapter 9 ( <a href="https://directives.sc.egov.usda.gov/">https://directives.sc.egov.usda.gov/</a> )
Precipitation events	ERA5Land ( <a href="https://climate.copernicus.eu/">https://climate.copernicus.eu/</a> )

Kc and CN values used in the model are shown in tables 4 and 5. Due to the lack of specific and detailed information about the LULC classes and their biophysical properties, average and standard values were selected for each class from recognised global data sources, following the recommendations of the

model developers. Kc values were obtained from the FAO guidelines (Allen et al., 1998). CN values for water bodies were set to 99 according to the model guidelines, for the other LULC classes values from the USDA Handbook (NRCS-USDA, 2007 Chap. 9) were used.

**Table 4.** Kc values used for LULC classes in the inVEST model.

LULC	Kc1	Kc2	Kc3	Kc4	Kc5	Kc6	Kc7	Kc8	Kc9	Kc10	Kc11	Kc12
UBAN	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4	0.4
BARE	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
AGRICULTURE	0.7	0.7	0.7	1.05	1.05	1.05	1.05	0.95	0.95	0.95	0.95	0.7
GRASSLAND	0.3	0.3	0.3	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75
SHRUBLAND	0.35	0.35	0.35	1	1	1	1	1	1	0.5	0.5	0.5
WATER	1	1	1	1	1	1	1	1	1	1	1	1
CONIFEROUS	1	1	1	1	1	1	1	1	1	1	1	1
FOREST	0.5	0.5	0.5	0.9	0.9	0.9	0.9	0.9	0.9	0.65	0.65	0.65

**Table 5.** CN values used for LULC classes in the inVEST model.

LULC	CN_A	CN_B	CN_C	CN_D
URBAN	98	98	98	98
BARE	77	86	91	94
AGRICULTURE	67	78	85	89
GRASSLAND	49	69	79	84
SHRUBLAND	55	72	81	88
WATER	99	99	99	99
CONIFEROUS	44	58	73	80
FOREST	30	55	70	77

The model generates 3 main outputs; quick flow (QF), local recharge (L) and base flow (B), more complete information on these outputs and a series of other intermediate products, as well as the methods to calculate each of them, can be found in the InVEST user manual (Sharp et al., 2019). To estimate the drought regulation ES, we used the generated local recharge map ( $L_i$ ) in mm, calculated according to:

$$L_i (mm) = P_i - QF_i - AET_i$$

where  $P_i$  is the annual precipitation;  $QF_i$  is the annual quick flow and  $AET_i$  is the current annual evapotranspiration, all of them for each pixel of the landscape. Although this model gives the results in mm, results must be interpreted as relative values (i.e. spatial indicators) and never as absolute values. Finally, the results obtained through this index have been incorporated into each drainage wing by calculating the average value. Moreover, the presence of forests plays an important role improving infiltration and water retention (Bruijnzeel, 2004; Ilstedt et al., 2016; Ilstedt et al., 2007), helping to generate a slower hydrological response of the catchment. To acknowledge this, we identified drainage wings dominated by natural forest (those with more than 60% natural forest) and regarded them as contributing effectively to the generation of the drought regulation ES (i.e. conservation of natural features), while the others were considered as failing to provide a satisfactory level of the ES. In this later drainage wings, we considered that it would be necessary to carry out specific actions to favor infiltration. Values of the local recharge index were classified according to percentiles 50 and 75 and the percentage of natural forest within the drainage wing. We obtained 3 classes of prioritization independently for both, con conservation and other actions.

## Thermal regulation

The thermal regulation ES is represented by the potential effect that the shadow cast by riparian forest might produce on diminishing the solar radiation incident in the river water. Overall, estimating the shadow cast by riparian forests on a river reach involves combining data on terrain elevation, riparian forest location, orientation of the river reach and sun angle and position. This estimation is made in a GIS environment by running several steps. First, we need collecting all necessary data for each DRN, including the (DEM), location and shape of riparian forests and sun angle (i.e. months for performing the calculation). By processing the DEM we generate a shaded relief map to be used to visualize terrain and identify areas that are likely to be shaded. Then, we conduct specific analysis to map the area occupied by riparian forests in order to identify its extent and averaged elevation. This information will be used to calculate the shadow cast by the trees. We need also to determine the angle and position of the sun during the time period of interest that will be in this case summertime (i.e. months of June, July and August). Finally, we create a map of the shadow cast by the riparian forest, which can be used to identify areas that are shaded and to plan management activities of restoration and conservation.

By grouping these steps in overall concepts, thermal regulation is performed in three stages:

1. Calculating **solar radiation** for each entire DRN using a DEM and DSM (Digital Surface Model) that involves elevating riparian forests where they have been mapped. A DSM is a digital representation of the surface of the earth and includes all features such as trees, buildings, and other above-ground structures.
2. Extracting solar radiation to **valley bottoms** with and without riparian forests elevation simulation.
3. Calculating the radiation reduction (thermal regulation) by **shadow cast of riparian forests** across valley bottoms.

### 1) Solar radiation calculation.

Solar radiation or insolation has been calculated based on methods from the hemispherical viewshed algorithm developed by Rich et al., (1994) and further developed by Fu & Rich, (2000) and Fu & Rich, (2002). The total amount of radiation calculated for a particular area is given as global radiation. It is calculated as the sum of direct and diffuse radiation of all sun map and sky map sectors, respectively. The calculation of direct, diffuse and global insolation are repeated for every location (i.e. pixel) on the topographic surface, producing insolation maps for the entire geographic area with the same resolution than the DEM (i.e. EUDEM of 25m pixel size).

### 2) Extracting solar radiation to valleys with and without riparian forests.

Most of natural and semi-natural ecosystems in Europe are exposed to a variety of threats, such as the intensification and industrialization of agricultural and forestry land uses, the abandonment of traditional farming activities or urban sprawl. Therefore, land use and land cover (LULC) mosaics are highly dynamic at varying spatio-temporal scales and need to be assessed for large territories and though time. At this regard, during recent decades, an increasing number of spatially-explicit methodologies have been developed to provide a better knowledge of past-to-present land cover changes at a regional scale. Many of these methods are based on remote sensing (RS) data and techniques for image classification and assessment, such as Landsat and Sentinel 2 imagery subjected to supervised classification based upon ground control and training datasets (Álvarez-Martínez et al., 2018; Alvarez Martinez et al., 2011), since they provide regional data at different temporal scales with low collection effort. Resulting maps allow defining LULC classes over a landscape matrix of different nature that will be driven, on their temporal dynamics, to former drivers of change such as climate dynamics and land use. In Europe, Corine (Coordination of Information on the Environment) Land

Cover maps are highly useful for assessing landscape structure across the whole territory and through time (Büttner et al., 2004). These maps provide detailed information about the different land cover types and their spatial distribution across the continent. Corine Land Cover (CLC) maps utilize a comprehensive and standardized classification system that covers a wide range of land cover types. This classification system allows for consistent and comparable analysis of land cover patterns and changes across different regions and countries in Europe. It provides a common language and framework for understanding the composition and arrangement of land cover types within the landscape, characterizing and monitoring the spatial arrangement and configuration of land cover types. In this work, we mapped LULC by refining CLC mosaics of the year 2018 by combining existing forest patches with NDVI values obtained from a time series of Sentinel 2 imagery for the years 2018 to 2020.

As explained previously, the calculation of ES provision in this work has been assessed by splitting the territory in a set of functional units. We include among them the valley bottom areas (i.e. floodplains; Benda et al., (2007)). The use of these functional units allows transferring digital information to the relevant spatial units in which the ES provision is generated. The delineation of the floodplain was calculated by using the NetMap software (Benda et al., 2007) using a geomorphological criterion based on the valley surface at a height of "n" times the bankfull depth of each river channel (Fernández et al., 2012). For this project, a multiplicative value of 5 was used as it is related to a flood of approximately 300-year return period (Ilhardt et al., 2000). The value of bankfull channel depth was estimated for each segment of the river network by means of a regional regression between drainage area, mean annual precipitation and field measurements of bankfull channel depth (for more detailed information, see Benda et al., (2011)).

Accurate information about ground elevation and canopy height are of great use in various scientific fields (Adam et al., 2020). Although heights can vary depending on location and environmental conditions, some studies have determined that riparian forests in Europe can range from 18.4 meters to 30.7 meters in height. Seidl & Lexer, (2013), analyzed data from 14 European countries and found that the average value for riparian forests in Europe ranged from 18.4 meters in Belgium to 30 meters in Norway. In this work, we used 20 meters height as an averaged estimate for riparian forests tree height across DRNs that fit better our study areas of interest. This value was added to the original DEM of 25 meters pixel size in those locations (i.e. pixels) across valley bottoms or floodplains with riparian forests, corresponding to broadleaved forests mapped for each DRN.

### **3) Radiation reduction (thermal regulation) by shadow cast of riparian forests.**

The calculation of the radiation reduction because of shadow produced by riparian forests is calculated by subtracting the original radiation calculated with the DEM with no vegetation elevation and with the DEM by adding 20-meter height to all pixels with mapped broadleaved forests (i.e. 20 meters height for forests mapped across valley bottoms). As a first stage, this calculation is made at the pixel level. Subsequently, data has been transferred to all functional units intersecting the valley bottoms defined below and corresponding to drainage wings, which are adjacent slopes draining into each river segment individually, and the functional units that divide the valley bottom areas.

The units of solar radiation are expressed in Watt-hours per square meter per day ( $\text{Wh/m}^2/\text{day}$ ), referring to the amount of energy that is received from the sun. The output for this ES is a raster dataset that represents the estimated amount of solar radiation received by each pixel in the study area, in units of  $\text{Wh/m}^2/\text{day}$  with values transferred to the functional units defined across the valleys.

## Carbon emissions

Carbon sequestration is a very important ES in the actual scenario of climate emergency (Le Quéré et al., 2009; Raymond et al., 2013). This ES is related to the amount of inorganic carbon that an ecosystem might be able to fix. However, carbon sequestration is rather difficult to be estimated in rivers, as biomass estimation and physiological rates of organisms within this very dynamic ecosystem are very difficult to be obtained for entire river networks. Having this in mind, we assume that carbon emissions from rivers have an organic control through the physiological rates of the many organisms inhabiting river ecosystems (Battin et al., 2023; Hotchkiss et al., 2015). We consider carbon emissions as an ecosystem process and not an ES, however, modelling carbon emissions to entire river networks is useful to infer the main drivers of carbon dynamics. The identification of these factors could cast light on the best management practices to reduce carbon emissions.

In this section, we describe the methodological steps followed to model the spatial and temporal dynamics of carbon emissions in the six DRNs during the three drying phases (pre-dry, dry and post-dry) using Dry CO<sub>2</sub> flux and Wet CO<sub>2</sub> flux data.

### CO<sub>2</sub> flux measurements

#### 1) Field sampling

For each DRN, site and sampling campaign, *in situ* CO<sub>2</sub> fluxes were determined in flowing waters and the dry riverbed (when present). For each habitat, measurements were made at 5 locations selected randomly within the river reach. CO<sub>2</sub> fluxes were measured with a floating or dry closed-loop chamber coupled to an infra-red gas analyzer (Picarro GasScouter™ G4301 Analyzer in Ain-Albarine and Vantaanjoki-Lepsämäenjoki, CO<sub>2</sub> and concentrations recorded every second; IRGA EGM-4 CO<sub>2</sub> Gas Analyzer in Guadairo-Genal and Morava-Velicka, CO<sub>2</sub> concentration recorded every 4.7 seconds; LGR-ICOS™ GLA131-GGA in Butižnica and Fekete-Bükkösd, CO<sub>2</sub> concentrations recorded every second). For each flux measurement, the gas concentration inside the chamber was recorded over 5 minutes. Under flowing conditions, the floating chamber was placed at the water surface, avoiding extremely turbulent areas to prevent its lifting and ensure the sealing. In dry conditions, a metal corer was used to insert a plastic collar (~2-5 cm depth) on which the dry chamber was placed. For dry riverbeds whose texture prevented the insertion of the metal corer, the collar was sealed to the dry surface using pottery clay to avoid any air exchange between the chamber and the atmosphere during the measure. Chambers were aerated until reaching the atmospheric gas concentrations in-between measures.

#### 2) CO<sub>2</sub> flux calculation

For each replicate measurement, the initial linear increase in CO<sub>2</sub> concentration over time was selected to calculate the CO<sub>2</sub> flux. This linear phase was first automatically identified by calculating the points with a significant change in slope ('*depseg*' function, R package *dpseg*) and then finally selected after a visual confirmation. A segment covering a minimum of 90 secs was always selected.

The CO<sub>2</sub> fluxes were calculated according to the following equation,

$$CO_2 \text{ flux} = \frac{\frac{dCO_2}{dt} * \text{MolecularMass} * \text{Pressure} * \text{Volume}}{\text{Area} * R * \text{AirTemp}}$$

where  $dCO_2/dt$  is the variation in gas concentration (ppm) over time  $t$  (in this case one hour); MolecularMass is the weight of C ( $g \text{ mol}^{-1}$ ); Pressure is the atmospheric pressure (atm) measured at each sampling site and campaign; Volume (L) is the total volume of the chamber plus the connection tubes and the internal cavity of the gas analyzer; Area is the surface of the chamber ( $m^2$ );  $R$  is the ideal

gas constant ( $L \text{ atm mol}^{-1} K^{-1}$ ) and AirTemp is the air temperature (K) measured at each site and campaign. For each campaign, site and habitat (flowing water, dry riverbed or isolated pool) we calculated the mean gas flux from the different replicates.  $CO_2$  flux was expressed in  $mg \text{ C m}^{-2} h^{-1}$ .

## **$CO_2$ extrapolation to the DRNs**

### **1) Wet and dry area calculation**

To transform  $CO_2$  flux data into absolute values per wet and dry area at each river segment, we estimated wet and dry width from daily discharge data from WP1. To estimate the hydraulic relationship between discharge and wet channel width we used a combination of the “downstream” (in space) and “at-a-station” (in time) formulations of hydraulic geometry of Leopold & Maddock, (1953), following the approach of Lamouroux & Souchon, (2002) and Morel et al., (2020). This equation identifies spatial variations in width (represented by the exponent  $bd$ ) and temporal variations (exponent  $b$ ), which is interesting when extrapolating width in space and time, because these two exponents generally differ (Leopold & Maddock, 1953),

$$\ln(WW) = \ln(ad) + bd * \ln(QM) + b * \ln\left[\frac{Q}{QM}\right]$$

Where  $WW$  is the wet width of the river segment,  $QM$  is the mean flow of 2021 and  $Q$  is the instantaneous flow. We fitted the equation to our field data with a linear regression ( $R^2 = 0.67$ ,  $p < 10^{-15}$ ) to extract the three parameters needed:  $ad = 6.48$ ,  $bd = 0.35$ ,  $b = 0.16$ . Then, the dry width was calculated as the difference between the bankfull width and the wet width of the segment. If wet width was higher than the bankfull, the dry width was considered zero.

### **2) $CO_2$ extrapolation and quantification**

We used Random Forest models to upscale the  $CO_2$  emissions (both from flowing water and dry riverbeds) to every single reach, and to the three selected periods (pre-dry season: march-april; dry-season: july-august; post-dry season: november-december). We fitted four RF models (flowing water perennial reaches, flowing water intermittent reaches, dry riverbed perennial reaches and dry riverbed intermittent reaches) on our field measured, with the predictor variables available in all DRN reaches to calculate the daily  $CO_2$  emission rates for each reach ( $mg \text{ C m}^{-2} d^{-1}$ ).

Those predictors included: daily discharge (only for flowing conditions, available from hydrological models in WP1), climatic variables (evapotranspiration and precipitation in a monthly scale, mean annual precipitation), topographical variables (mean elevation and reach slope), landcover (proportion of broadleaf, coniferous forests, agricultural area and grassland in the upstream river catchment) variables indicating the position within the river network (distance to the water source, upstream riverlength) and drying metrics (only for intermittent reaches, flow prevalence and drying severity during 2021). To avoid overfitting, model simplification was performed by repeatedly removing the least important variable from the model and recalibrating it until the model with lowest root mean square error (RMSE) was found.

The models for flowing conditions explained 51 and 32 % of the variance for perennial and intermittent reaches respectively. The models for dry riverbeds explained 56 and 70 % of the variance for perennial and intermittent reaches respectively.

A river emits inorganic carbon into the atmosphere if the sum of external and internally produced  $CO_2$  is higher than the  $CO_2$  fixated into organic carbon. Additionally, intermittent rivers present another component as they dry out completely during parts of the year, interrupting the aquatic carbon

processes. Total CO<sub>2</sub> flux (g C m<sup>-2</sup> d<sup>-1</sup>) from a river that suffers wetted channel contraction can be expressed as:

$$\text{Total CO}_2 \text{ flux} = \text{Wet CO}_2 \text{ flux} + \text{Dry CO}_2 \text{ flux}$$

Where the Dry CO<sub>2</sub> flux (g C m<sup>-2</sup> d<sup>-1</sup>) is the dry sediment respiration. The Wet CO<sub>2</sub> flux (g C m<sup>-2</sup> d<sup>-1</sup>) originates from internal production and external inputs.

## Results and discussion

We present below the main results obtained when modeling the ecological functions controlling each of the ES selected as well as the resulting maps of ES potential provision at the landscape unit level, disentangling which areas may correspond to conservation or restoration when applying environmental management actions at the watershed or riparian networks levels.

### Water provisioning

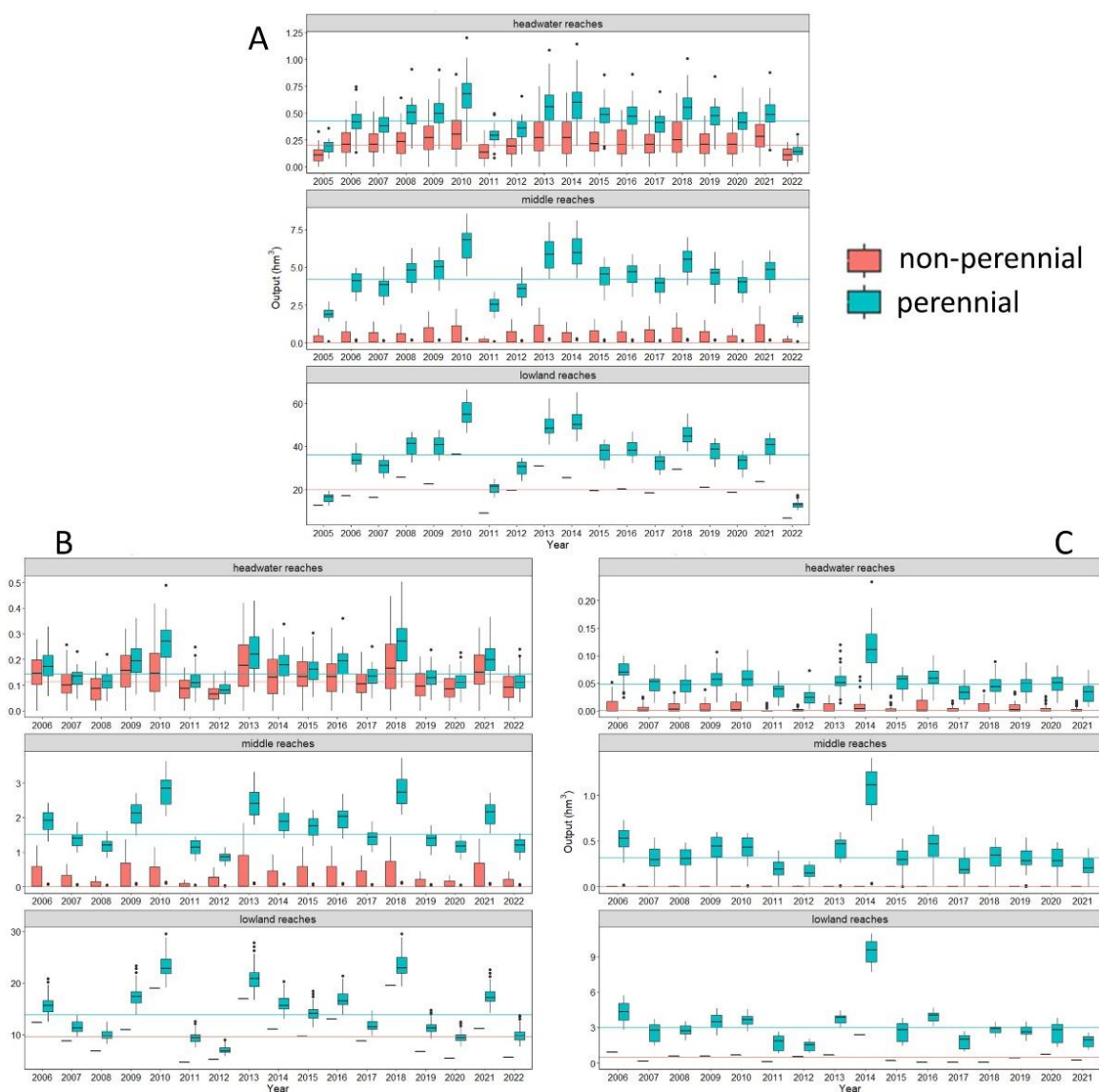
The procedure to select river reaches (Steps 1-3) where water provisioning was further assessed (Step 4) showed that headwater reaches ranged between 0.8 and 1.6 km<sup>2</sup>, middle reaches ranged between 9.9 and 19.5 km<sup>2</sup> and lowland reaches ranged between 63.6 and 233 km<sup>2</sup> (Table 6). This highlights a good agreement between DRNs, except for the larger size class, as it basically depends on the area close to the outlet of the DRN catchment which varied from one DRN to another. As expected, the number of Non-Permanent Reaches (NPR) was usually greater than the number of PR reaches in the headwater class of Krka-Butišnica, Ain-Albarine and Guadiaro-Genal, while the opposite result was observed in Morava-Velička, Fekete-Bükkösdi and Vantaanjoki-Lepsämäjoki. In the latter case, this distribution of PR and NPR might be due to a lower number of NPR reaches in the DRN, as it is located in the highest latitude where intermittency is less common. By contrast, the result obtained for the Morava-Velička and Fekete-Bükkösdi DRNs might be due to anthropogenic factors (e.g. irrigation lands in Velička) or errors in the delineation of river network and hydrological modelling accuracy (see Künné et al., 2022). Most of the reaches selected in the middle reaches were PR, while only the Krka-Butišnica, Ain-Albarine and Fekete-Bükkösdi, still included a sufficient number of NPR reaches (n≥3) to analyze the water provisioning. By contrast, none of the selected reaches in the lowland class comprised sufficient number of NPR reaches (Table 6).

**Table 6.** Catchment area of the selected reaches according to the three size classes (headwater, Middle and lowland reaches) in each DRN and number of permanent (PR) and non-permanent (NPR) reaches in each size class and DRN.

DRN	Headwater (p25)			Middle (p75)			Lowland (p90)		
	Km <sup>2</sup>	PR	NPR	Km <sup>2</sup>	PR	NPR	Km <sup>2</sup>	PR	NPR
Krka-Butišnica	0.8 ±0.2	24	34	7.6 ±1.5	23	3	63.6 ±12.7	21	1
Morava-Velička	1.6 ±0.3	12	7	19.5 ±3.9	8	1	76.5 ±15.3	11	0
Vantaanjoki-Lepsämäjoki	0.8 ±0.2	26	4	49.9 ±2	19	0	64.7 ±12.9	16	0
Ain-Albarine	1.3 ±0.3	1	38	15.0 ±3	12	4	140 ±28	13	0
Fekete-Bükkösdi	1.4 ±0.3	25	0	11.4 ±2.3	7	8	107 ±21.4	6	0
Guadiaro-Genal	1.2 ±0.2	0	37	13.5 ±2.7	8	2	233 ±46.6	15	0

The analysis of the annual water availability (i.e., water that can be provisioned) in the Krka-Butišnica DRN showed median values of 0.43, 4.22 and 36.06 hm<sup>3</sup> for the headwater, middle and lowland PR,

respectively (Figure 3A). These amounts showed important reductions in the NPR. In the headwater reaches the available water was cut by half ( $0.2 \text{ hm}^3$ ), while in the middle reaches it is practically null (Table 7). In the case of NPR in lowland reaches, the selection procedure only considered one reach. Assuming a low representative of this selection, we omitted further analysis of the results in this reach.



**Figure 3.** Box-plots showing the ES water provisioning in Krka-Butižnica DRN (Croatia) considering the amount of water for a) the whole year; b) the wet season (January-February-March); c) the dry season (July-August-September). For each year of the series, the boxes represent the 25th and 75th percentiles, the middle line within the box represents the median and the whiskers represent the 10th and 90th percentiles of the distribution of values of the perennial and non-perennial river reaches in each catchment area class (headwater, middle and lowland reaches). The solid lines in each graph represents the median value of the whole time series of the perennial and non-perennial river reaches in each catchment area class.

Water provisioning in the PR reached median values of 0.15, 1.5 and  $13.91 \text{ hm}^3$  during the wet season (Figure 3B) and median values of 0.05, 0.32 and  $2.97 \text{ hm}^3$  during the dry season in the headwaters, middle and lowland reaches, respectively (Figure 3C). This means that the water provisioning in the PR during the wet season attained 35% of the total annual amount of water available during the wet season, while it accounted for 11.5%, (headwaters) 7.5% (middle) and 8.2% (lowland) of the total annual balance in the PR. The Krka-Butižnica DRN showed the smallest differences in the water provisioning between the dry and wet seasons compared with the other 5 DRNs. Smaller differences might be related to the presence of storage mechanism in the DRNs, such as aquifers, ponds, wetlands

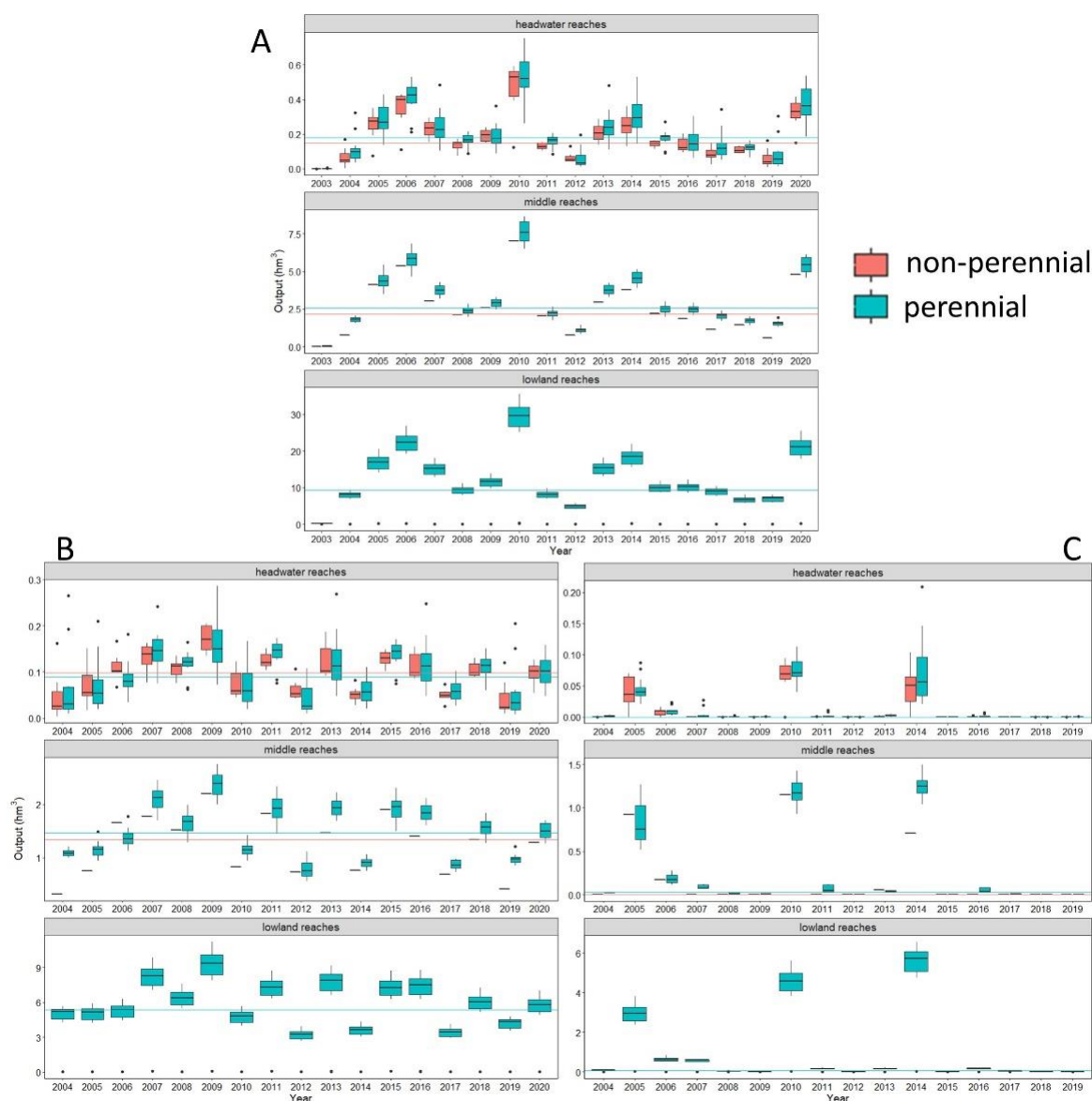
or well-preserved mature forest (Belmar et al., 2018; Mcnamara et al., 2011). However, it must be also pointed out that the small disparity between dry and wet season was only observed in the PR. In the NPR, the 55% of the water in NPR located in headwaters is provisioned during the three months of the wet season ( $0.11 \text{ hm}^3$ ) while this amount is reduced below 0.3% during the dry season ( $<0.05 \text{ hm}^3$ ).

**Table 7.** Summary of the ES water provisioning results. Median values of the water availability considering the whole year (annual) and the wet and dry seasons for the headwater, middle and Lowland perennial (PR) and non-perennial reaches (NPR) in the six DRNs. Empty cells mean no reaches in this size class. Light gray indicates low number reaches ( $<3$ ) in this size class.

	Krka-Butišnica		Morava-Velička		Vantaanjoki-Lepsämäenjoki		Ain-Albarine		Fekete-Bükkösdi		Guadiaro-Genal	
	PR	NPR	PR	NPR	PR	NPR	PR	NPR	PR	NPR	PR	NPR
<b>Annual</b>												
Headwater	0.43	0.20	0.18	0.15	0.24	0.23	0.76	1.06	0.07			0.24
Middle	4.22	0.00	2.58	2.17	2.44		8.29	12.1	0.66	0.14	3.33	3.24
Lowland	36.06	19.84	9.32		16.21		134.3		3.20		74.23	
<b>Wet</b>												
Headwater	0.15	0.11	0.09	0.10	0.07	0.07	0.32	0.40	0.03			0.12
Middle	1.51	0.00	1.46	1.34	0.75		3.70	4.90	0.36	0.07	1.71	1.59
Lowland	13.91	9.64	5.35		5.15		51.69		1.87		36.21	
<b>Dry</b>												
Headwater	0.05	0.00	0.00	0.00	0.02	0.01	0.02	0.03	0.00			0.00
Middle	0.32	0.00	0.02	0.00	0.19		0.07	0.20	0.03	0.00	0.00	0.00
Lowland	2.97	0.47	0.06		1.06		4.73		0.21		0.38	

The analysis of the annual water availability in the Morava-Velička DRN showed median values of 0.18, 2.58 and  $9.32 \text{ hm}^3$  for the headwater, middle and lowland PR, respectively (Table 7; Figure 4A). Differences between PR and NPR were less pronounced than in other DRNs, revealing differences around 20% ( $0.15 \text{ hm}^3$  in headwater NPR; Table 7). Moreover, it can be noted that in several years of the series the differences between water provisioning in PR and NPR was negligible. Results also pointed out significant inter-annual differences, i.e. differences between years in the three-size class reaches. In this regard, we observed that for some years (e.g. 2006, 2010 or 2020) water provisioning can almost double the median annual value of the series, while other years (e.g. 2003, 2012) this ES can be seriously reduced (Figure 4A) taken values close to zero.

Finally, analyses also showed significant intra-annual differences. The wet season provided, at least, 50% of all available annual water (49.3, 56.6 and 57.4% for the headwater, middle and lowland PR, respectively, Figure 4B), while the dry season provided less than 1% in the three size-classes (Table 7; Figure 4C). This observation is even more pronounced in headwater NPR, where wet seasons provide more than 65% of the total annual water budget ( $0.1$  out of  $0.15 \text{ hm}^3$ ). This unbalance might be related with the low regulatory capacity of the catchment, which boost a rapid drainage of water from the wings to the river network through surface flow, which usually reduce the provision of water during the dry season. In this case, the low regulatory capacity might be related to the large agricultural areas. This can have dramatic consequences for the water uses that depend of the water availability in both PR and NPR and represent a critical environmental and social issue that must be considered in water planning. In this regard, investment in nature-based solutions, such as blue and green infrastructures networks, would suppose an adequate strategy to improve the regulatory capacity of the catchment (Staccione et al., 2021).

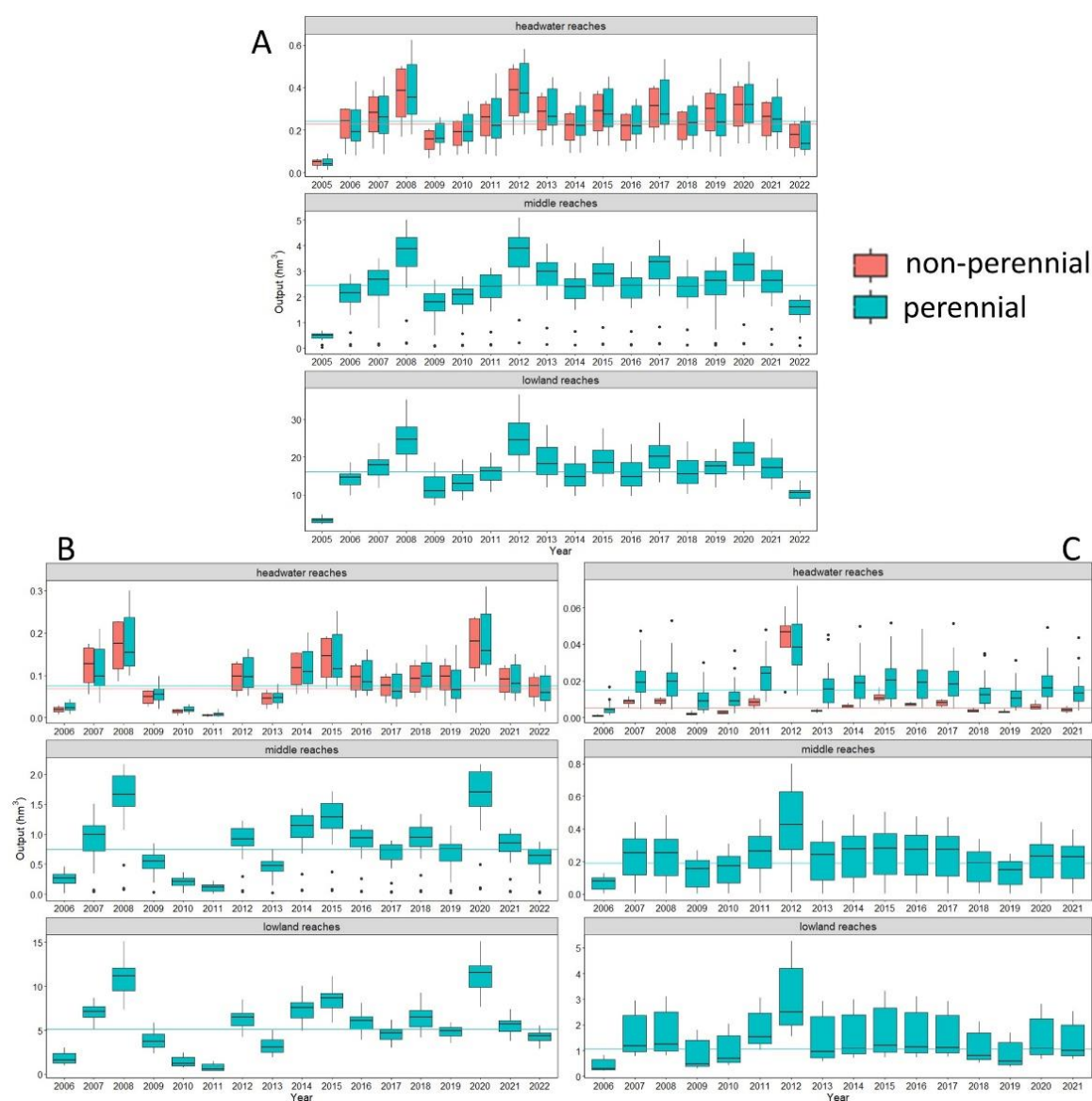


**Figure 4.** Box-plots showing the ES water provisioning in Morava-Velička DRN (Czech Republic) considering the amount of water for a) the whole year; b) the wet season (January-February-March); c) the dry season (July-August-September). For each year of the series, the boxes represent the 25th and 75th percentiles, the middle line within the box represents the median and the whiskers represent the 10th and 90th percentiles of the distribution of values of the perennial and non-perennial river reaches in each catchment area class (headwater, middle and lowland reaches). The solid lines in each graph represents the median value of the whole time series of the perennial and non-perennial river reaches in each catchment area class.

The annual water availability in the Vantaanjoki-Lepsämäenjoki showed median values of 0.24, 2.44 and 16.21  $\text{hm}^3$  for the headwater, middle and lowland PR, respectively (Figure 5A). According to the number of reaches in each size class in this DRN (Table 7), comparison of the water provisioning between PR and NPR were only possible for headwaters. In this regard, results pointed out that the water provisioning in headwater PR and NPR was practically equivalent. This pattern was observed also in the independent analysis of the wet season (Figure 5B) while for the dry one, water availability was cut by half (0.07  $\text{hm}^3$  and 0.01  $\text{hm}^3$  in the wet and dry seasons, respectively; Figure 5C and Table 7).

Water provisioning for the wet season represented around 30% of the total annual budget (0.07, 0.75 and 5.15  $\text{hm}^3$  for the headwater, middle and lowland PR; Figure 5B) while this value was reduced below 8% for the dry season in all the size classes (0.02, 0.19 and 1.06  $\text{hm}^3$  for the headwater, middle

and lowland PR; Figure 5B). Moreover, this DRN showed that even the NPR during the dry seasons still were able to provision 2% of the total annual water budget. This result is not surprising since it is the northern most catchment of DRyVER study area. In this regard, these small differences might be related to a less marked seasonality in the distribution of annual rainfall associated with the influence of the subarctic climate in this boreal DRN, i.e. less summer aridity, and also to snowfall and snowmelt processes, which are determinant in the hydrological pathways and travel times in northern Europe landscapes (Jutebring Sterte et al., 2021). Moreover, as it was argued for the Krka-Butižnica DRN, this annual stability could also be related with the larger presence of water storage mechanism, such as lakes and wetlands, in the boreal region (Sundseth K, 2010). The presence of such natural elements could also be the responsible for the small inter-annual differences in the annual water budget observed, which were less pronounced than in other DRNs. Nonetheless, analysis highlighted contrasting results along the year. For instance, it can be observed that some years (e.g. 2006, 2010 or 2011) the water provisioning was significantly lower than the median value of the series during the wet season (Figure 5B).



**Figure 5.** Box-plots showing the ES water provisioning in Vantaanjoki-Lepsämäenjoki DRN (Finland) considering the amount of water for a) the whole year; b) the wet season (January-February-March); c) the dry season (July-August-September). For each year of the series, the boxes represent the 25th and 75th percentiles, the middle line within the box represents the median and the whiskers represent the 10th and 90th percentiles of the distribution of values of the perennial and non-perennial river reaches in each catchment area class (headwater, middle and lowland reaches). The solid lines in each

graph represents the median value of the whole time series of the perennial and non-perennial river reaches in each catchment area class.

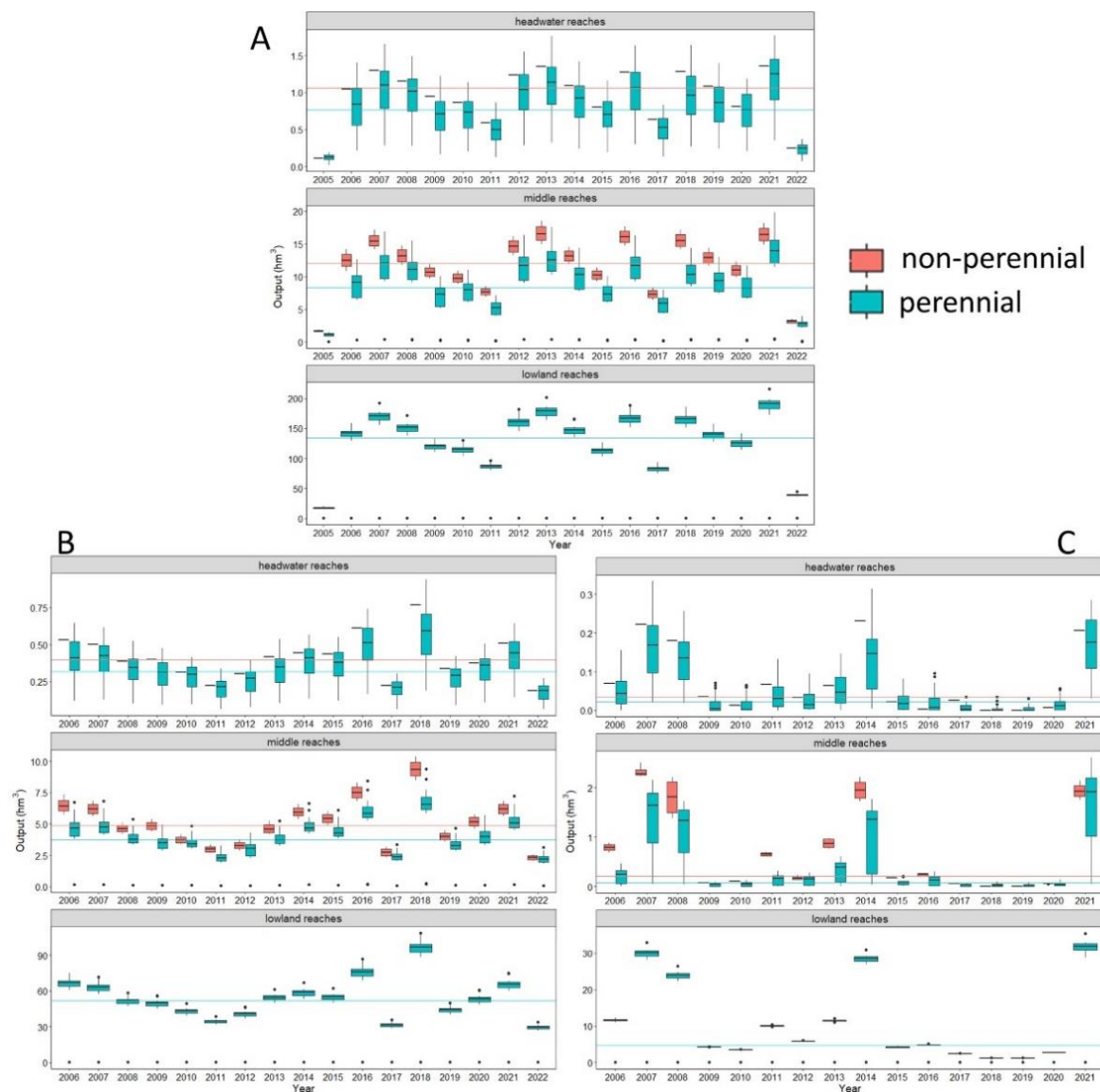
The annual water availability in the Ain-Albarine DRN showed median values of 0.76, 8.29 and 134.3  $\text{hm}^3$  for the headwater, middle and lowland PR, respectively (Table 7; Figure 6A). It must be pointed out that this DRN showed the highest values of the ES water provisioning in the three size-classes PR compared with other DRNs, while it did not account with the largest catchment areas (Table 6), meaning 1) a larger amount of annual precipitation and/or 2) the configuration of the Ain-Albarine DRN in terms topography, land uses and geology that potentially generates a more “efficient” transferability of precipitation to runoff and discharge (Graf & Lecce, 1988; Price, 2011). The other critical issue observed in this DRN is that for both, headwater and middle reaches, water provisioning in the NPR were larger than in the PR (Table 7; Figure 6). This contradicts initial expectations, according to which PR should provide more water than NPR. In headwaters this contradiction might be due to the low number of PR selected in headwater ( $n=1$ ; Table 6). However, in the case of middle reaches, and assuming that results from the hydrological model are satisfactory (Künne et al., 2022), this result might be associated to specific configuration and characteristics of this DRN that should be reviewed carefully to understand the hydrological process that is driving this pattern.

As it was observed in the other DRNs the water available in the wet season provided around 40% of the annual water budget in the PR (3.70 and 51.69  $\text{hm}^3$  for the middle and lowland PR) and in the NPR (0.4 and 4.90  $\text{hm}^3$  for the headwater and middle NPR). By contrast, the available water during the dry season is seriously reduced, accounting with less than 4% in all cases, i.e. middle and lowland PR and headwater and middle NPR (Table 7).

Finally, results also highlighted a significant inter-annual variability, which is most noticeable in the middle PR and NPR, and especially during the dry season (Figure 6C). While in the PR there is an important variability between reaches, it can be observed that for some years (e.g. 2007, 2008, 2014 and 2021) water provisioning during this season can be multiplied by 10 respect the median value of the series.

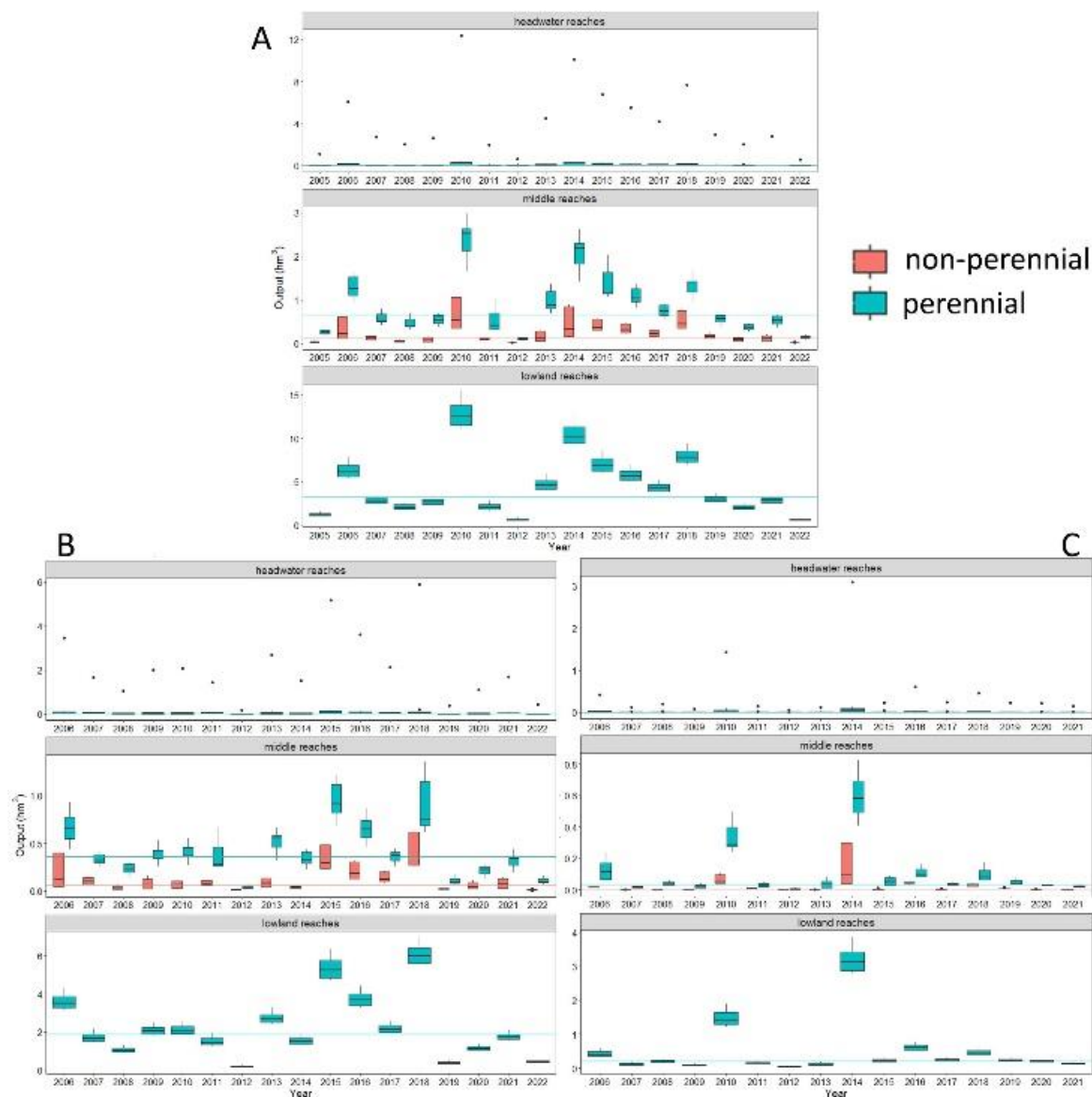
The annual water availability in the Fekete-Bükkösi DRN showed median values of 0.07, 0.66 and 3.2  $\text{hm}^3$  for the headwater, middle and lowland PR, respectively (Table 7; Figure 7A). This DRN showed a particularity, since the reach selection procedure did not select any NPR in the headwater reaches and, by contrast, selected a good representation of NPR in the middle reaches ( $n=7$  out of 15). Moreover, it must be also noted that, even though we did not find any NPR reach in the headwater reaches, in most of the cases, the water provisioning was practically null in this class. These results might be associated with hydrogeological features governing losses (sinks) of surface water to hyporheic water pathway through the middle reaches, but also to poor hydrological modelling results obtained in this DRN.

Irrespectively, we observed significant reduction of the annual water provisioning in NPR respect the PR in the middle reaches, showing a reduction close to 80% respect the RP in the same size class (Table 7; Figure 7A and B).



**Figure 6.** Box-plots showing the ES water provisioning in Ain-Albarine DRN (France) considering the amount of water for a) the whole year; b) the wet season (January-February-March); c) the dry season (July-August-September). For each year of the series, the boxes represent the 25th and 75th percentiles, the middle line within the box represents the median and the whiskers represent the 10th and 90th percentiles of the distribution of values of the perennial and non-perennial river reaches in each catchment area class (headwater, middle and lowland reaches). The solid lines in each graph represents the median value of the whole time series of the perennial and non-perennial river reaches in each catchment area class.

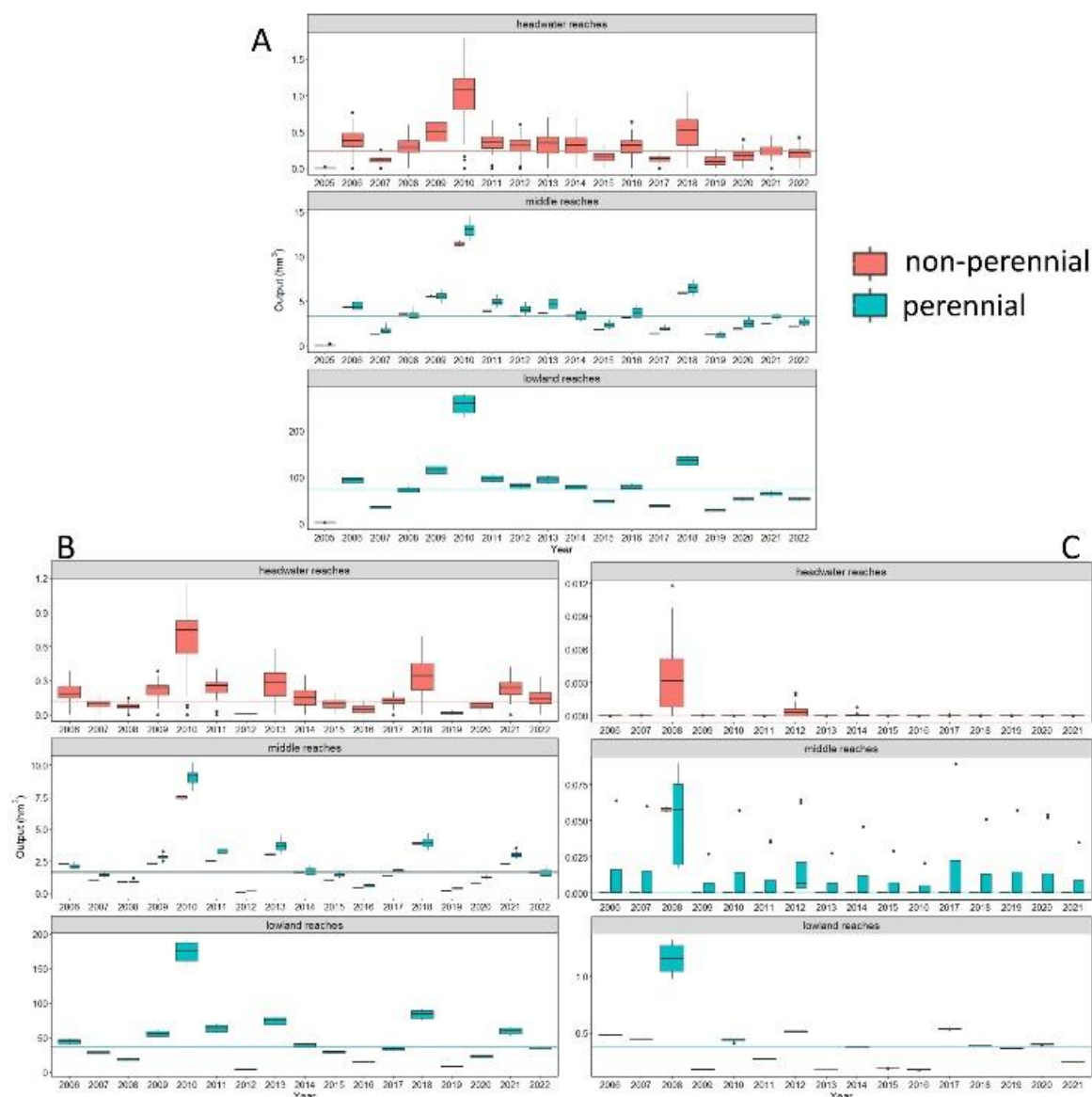
Water provisioning in the PR reached median values of 0.03, 0.36 and 1.87 during the wet season (Figure 7B) and median values of 0.005, 0.03 and 0.21  $\text{hm}^3$  during the dry season in the headwater, middle and lowland reaches (Figure 7C). This means that the water provisioning in the PR during the wet season supposed almost 50-60% of the total annual amount of water available, while during the dry season this percentage reached over 5% of the total annual balance (Table 7; Figure 7B and C).



**Figure 7.** Box-plots showing the ES water provisioning in Fekete-Bükkösi DRN (Hungary) considering the amount of water for a) the whole year; b) the wet season (January-February-March); c) the dry season (July-August-September). For each year of the series, the boxes represent the 25th and 75th percentiles, the middle line within the box represents the median and the whiskers represent the 10th and 90th percentiles of the distribution of values of the perennial and non-perennial river reaches in each catchment area class (headwater, middle and lowland reaches). The solid lines in each graph represents the median value of the whole time series of the perennial and non-perennial river reaches in each catchment area class.

The annual water availability in the Guadiaro-Genal DRN showed median values of 3.33 and 74.23  $\text{hm}^3$  for middle and lowland PR, respectively and median values of 0.24 and 3.34  $\text{hm}^3$  for the headwater and middle NPR (Table 7; Figure 8A). Attending to the location of the Guadiaro-Genal DRN, i.e., the southernmost DRN of DRYVER and consequently the most arid one, we should not expect such large water provisioning values compared with other DRNs. In this regard, Guadiaro-Genal showed the second highest annual water budget after Ain-Albarine (Table 7; Figure 8A). Another unexpected result in Guadiaro-Genal arose from the comparison of water provisioning between PR and NPR. This comparison was only possible for middle reaches and suggested that there were not differences in the water provisioning of PR and NPR both at the annual or seasonal scale. This results clearly contradicts our initial expectation, especially in the most arid catchment of the DRYVER. Both results might be due to the lack of proper hydrological records inputs to calibrate the JAMS/J2000 modeling

system/hydrological model that have probably translated to inaccurate discharge simulations (Künne et al., 2022).



**Figure 8.** Box-plots showing the ES water provisioning in Guadiaro-Genal DRN (Spain) considering the amount of water for a) the whole year; b) the wet season (January-February-March); c) the dry season (July-August-September). For each year of the series, the boxes represent the 25th and 75th percentiles, the middle line within the box represents the median and the whiskers represent the 10th and 90th percentiles of the distribution of values of the perennial and non-perennial river reaches in each catchment area class (headwater, middle and lowland-reaches). The solid lines in each graph represents the median value of the whole time series of the perennial and non-perennial river reaches in each catchment area class.

On the other hand, it must be noted that during the dry season the water provisioning values are drastically reduced compared with the annual and wet season budgets (Table 7; 8C), which actually agreed with the expected hydrological patterns of an arid catchment. Moreover, Guadiaro-Genal showed the highest differences between the wet and the dry season compared with the other DRNs, independently of the type or size of river considered. In all the cases, the water provisioning during the three months of the wet season supposed one half of the total annual budget, while the availability of water during the summer months was practically negligible (<0.1%). This disparity might be associated with the very scarce precipitations events in of the Mediterranean climate, but also with the absence of natural storage mechanism in the catchment. This has important implication for water uses and water resources management, and is a critical issue considering that the severity of droughts

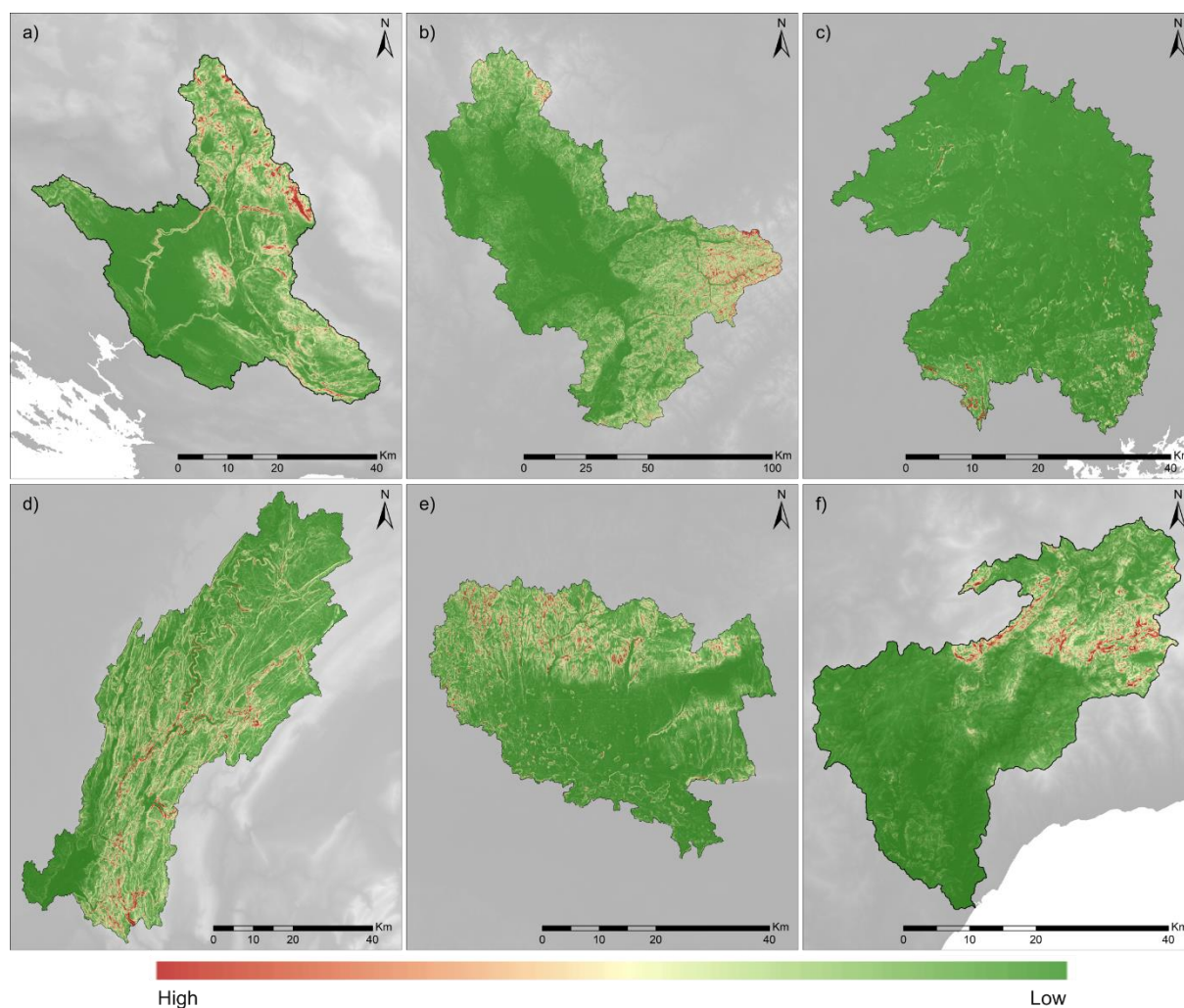
is expected to worsen with climate change in the southern most arid regions (Spinoni et al., 2021). Hence, the investment in nature-based solutions, such as increase of natural forest area, recovery of floodplains or restoration of ponds and wetlands should be prioritize to face future water security challenges in these regions (Cassin & Matthews, 2021).

## Flooding regulation

### Flooding regulation in slopes

The Hydrological Regulation Index (HRI), which is an estimation of the potential capacity of a catchment to generate runoff, is shown in the Figure 9. The spatial pattern occurs in the same way for all DRNs in the DRYvER project, where the combination of high values for each of the evaluated factors leads to higher index values, i.e. areas with the steepest gradients, shortest times and highest precipitation. High index values mean areas where more surface runoff is generated and less hydrological regulation is happening.

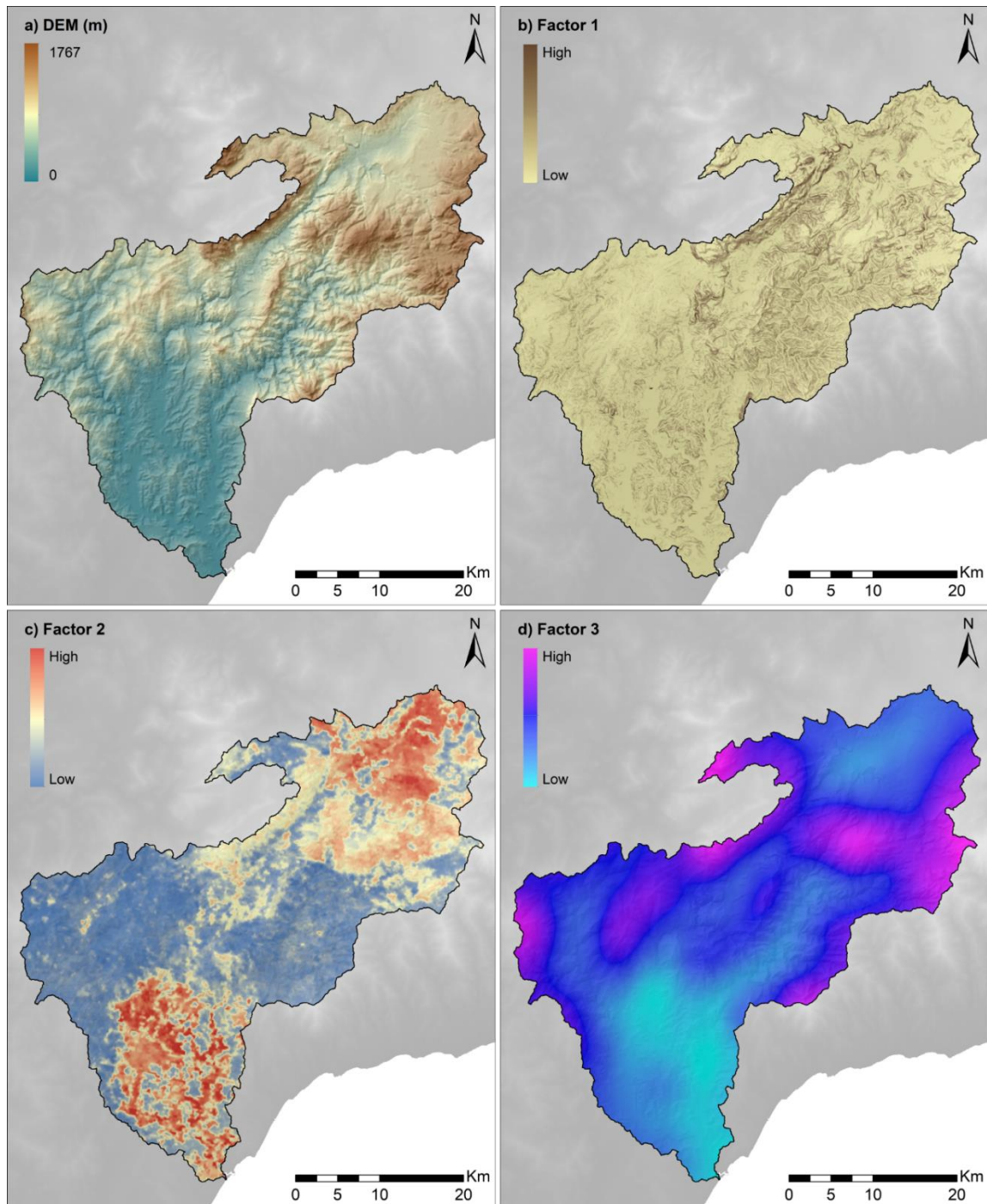
The catchments of Krka, Ainand Fekete have a larger area at risk of generating surface runoff compared to the other case studies. Morava shows the highest risk in specific areas located in the eastern part of the basin. In the case of Vantaanjoki, the predominance of low values of the HR index across the catchment is particularly remarkable, which is probably related to the flatness of the catchment. The Guadiaro catchment concentrates the highest risk of runoff generation in the upper part of the basin.



**Figure 9.** Hydrological Regulation Index (HR) in slopes in a) Croatia – Krka, b) Czech Republic – Morava, c) Finland – Vantaanjoki, d) France – Ain, e) Hungary – Fekete and f) Spain – Guadiaro. ES to regulate the hydrological response (runoff) for all the DRNs belonging to the DRYvER project.

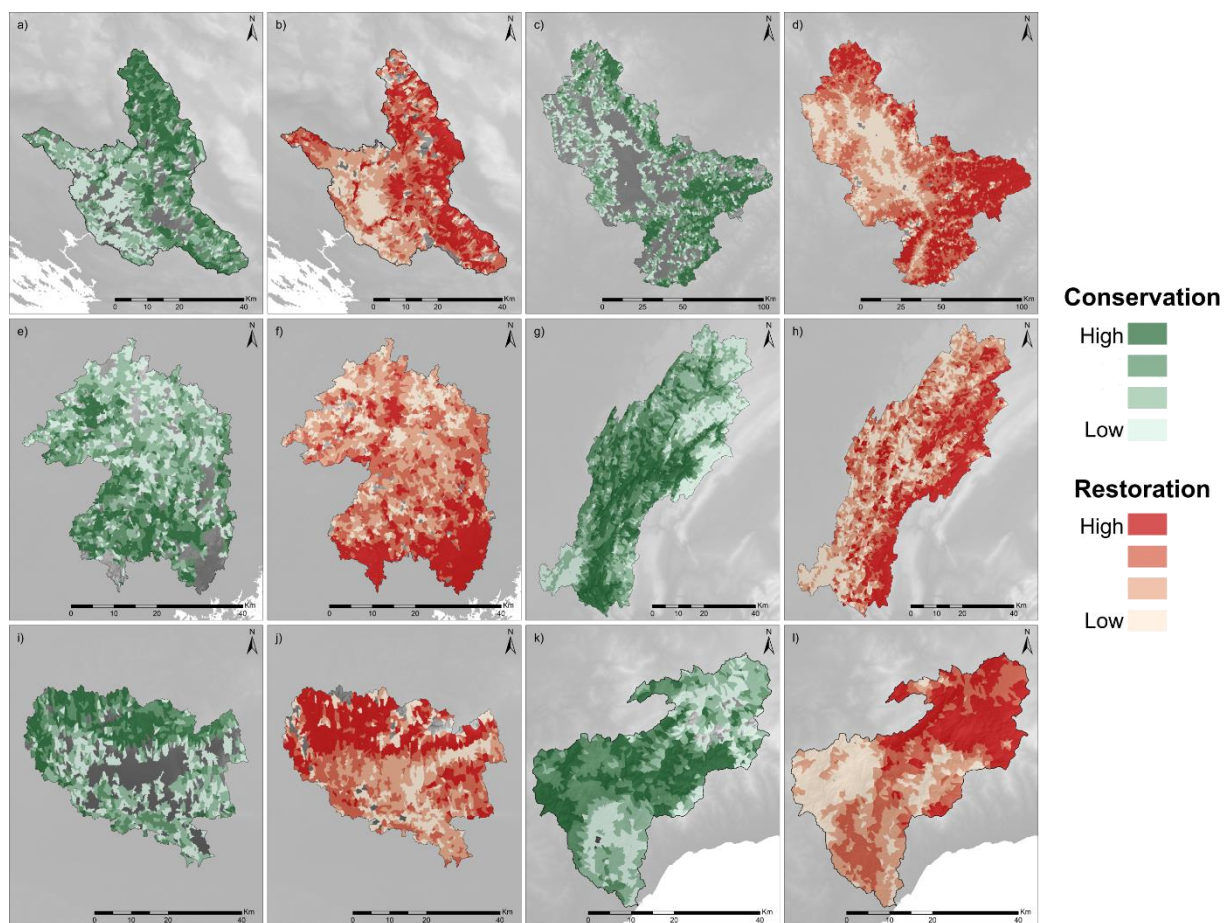
The case study of Spain, the Guadiaro catchment, which has a rougher relief distributed throughout the basin, is interesting for a more in-depth analysis of the index generated and the variables involved (Figure 10), serving as an illustration to better understand the results obtained. As mentioned, the

Guadiaro catchment has a heterogeneous orography (Figure 10a), reaching altitudes of 1767 m. Higher values of factor 1 (figure 10b) are found in areas with steeper slopes and where saturation of the first 30 cm of soil has occurred. Regarding the soil hydraulic properties evaluated through the inverse of the lateral flow travel time (factor 2), the areas with higher values (in red) are shown in Figure 10c, meaning shorter travel times and higher velocities. Finally, precipitation (factor 3) in the catchment varies from 677 to 890 mm/year and is spatially distributed as is shown in Figure 10d, where the lowest rainfall occurs in the lower altitude areas and valley bottoms. The higher values of the index, restricted to the upper part of the Guadiaro catchment, are mainly determined by the combination of higher rainfall and higher velocities.



**Figure 10.** Related factors of the Hydrological Regulation Index for the Guadiaro catchment; a) Digital Elevation Model (DEM); b) Factor 1 slope and soil saturation; c) Factor 2: inverse of lateral flow travel time; d) Factor 3: precipitation.

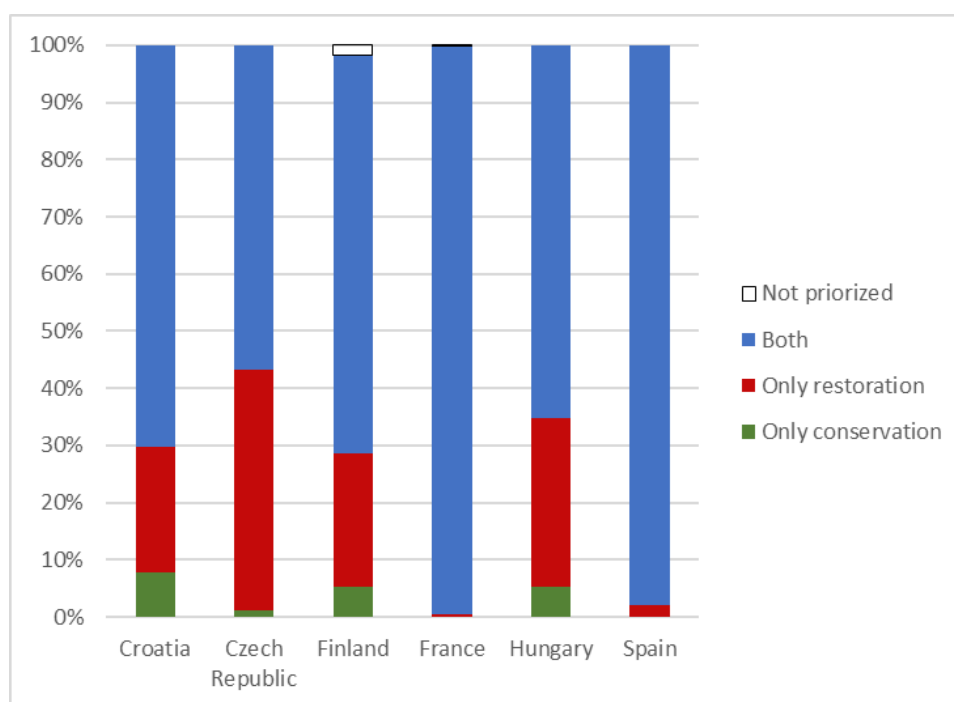
Based on the introduction of the forest in the HR index, the proposed areas to be conserved or restored are classified into four categories according to the degree of priority for action (Figure 11), i.e. those in which there is a greater interest in action because they generate a greater benefit as ES.



**Figure 11.** Proposed conservation (green) and restoration (red) drainage wings for the flooding regulation in slopes in a-b) Croatia – Krka , c-d) Czech Republic – Morava , e-f) Finland – Vantaanjoki , g-h) France – Ain , i-j) Hungary – Fekete and k-l) Spain – Guadiaro.

Figure 12 shows the proportion of functional units for each type of action recommended when talking about the ES of runoff regulation in slopes. There are drainage wings where only conservation actions are proposed (in green), only restoration (in red), others where both types of action are recommended (in blue) and finally, others where no action of any kind is necessary (only in the case of Vantaanjoki, 2%). In more than 50% of the drainage wings of all case studies, both types of action are required at the same time, with France and Spain being the most affected, probably due to their rougher orography.

The presence of mature forests in areas where we find higher HR values has a positive effect on regulating the hydrological response of the catchment, due to their "sponge" character (Belmar et al., 2018; Peña-Arancibia et al., 2019), leading to a slower response. Slow hydrological responses are related to slower flow velocities, higher flow concentration times and higher water retention, which generates flood protection. On the contrary, the absence of forest in these areas is the most unfavourable scenario for the hydrological response of the catchment, especially when it happens in areas where rainfall is higher and permeability is low.



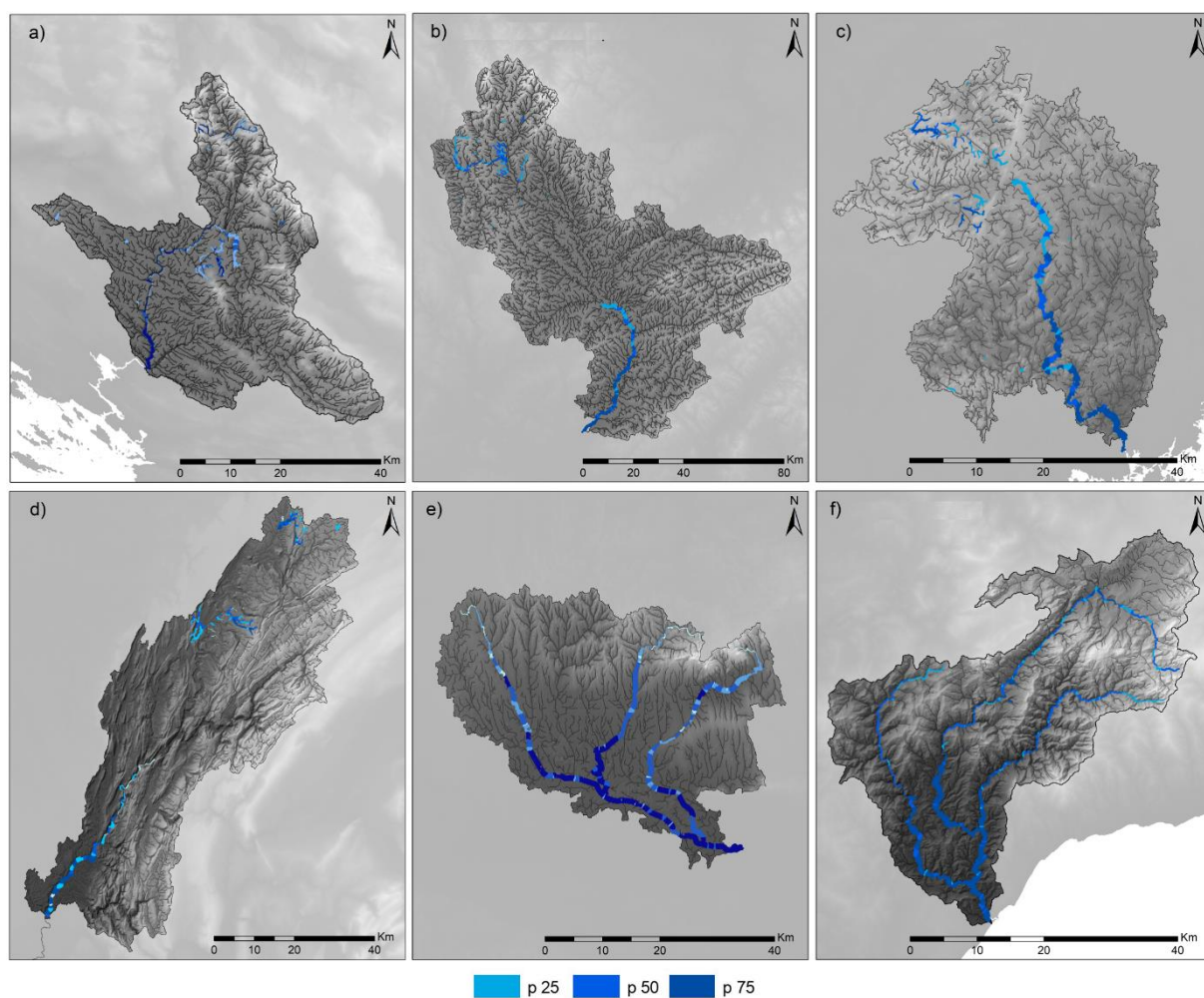
**Figure 12.** Proportion of prioritized drainage wings in terms of hydrological regulation in slopes. DRNs: Croatia, Czech Republic, Finland, France, Hungary and Spain.

### Water storage in floodplains

We assessed the hydrological response in floodplains through the potential water storage capacity in the event of a typical flood flow that overflows the channel. The volume of stored water is defined from a reference plane below which water remains temporarily stored.

Figure 13 shows the plains with the greatest storage capacity (p 50 and p75) are, in general, located in the middle and lowland areas of the basins. Therefore, the plains adjacent to the middle section of the basin are those with the greatest flood abatement potential, followed by the lower areas, closer to the mouth of the river.

As shown above, the Krka, Fekete and Guadiaro basins (Figure 13a, 13e and 13f) have the highest number of plains in the mouth with a value p 75, while the Morava, Vantaanjoki and Ain rivers basin (Figure 13b, 13c and 13d) have the highest number of plains with a value p 50. The Krka, Vantaanjoki, Fekete-and Guadiaro river basins (Figure 13a, 13c, 13e and 13f) have the highest number of plains in the middle section of the basin, with higher values in Guadiaro and Fekete than in the other two. In the Morava and Ain rivers basin (Figure 13b and 13d). In the catchment of the Morava and Ain rivers (Figure 13b and 13d), some floodplains are observed in the headwater area of the catchment with low capacity values (p 25), While in the Krka river basin (Figure 13a), no flood plains are visible in the headwater of the basin. Finally, in the basins of the Vantaanjoki, Fekete, and Guadiaro rivers, there are plains with accumulation capacity in all sections of the basin connected to the main river in the case of the Vantaanjoki basin (Figure 13c), and to the three main rivers in the Fekete and Guadiaro basins (Figure 13e and 13f). All this information allows us to determine that the Guadiaro and Fekete river basins have the greatest protection against flooding on the plains, followed by the Vantaanjoki and Ain river basins, and finally, the Krka and Morava river basins, which have the least accumulation capacity on their plains.

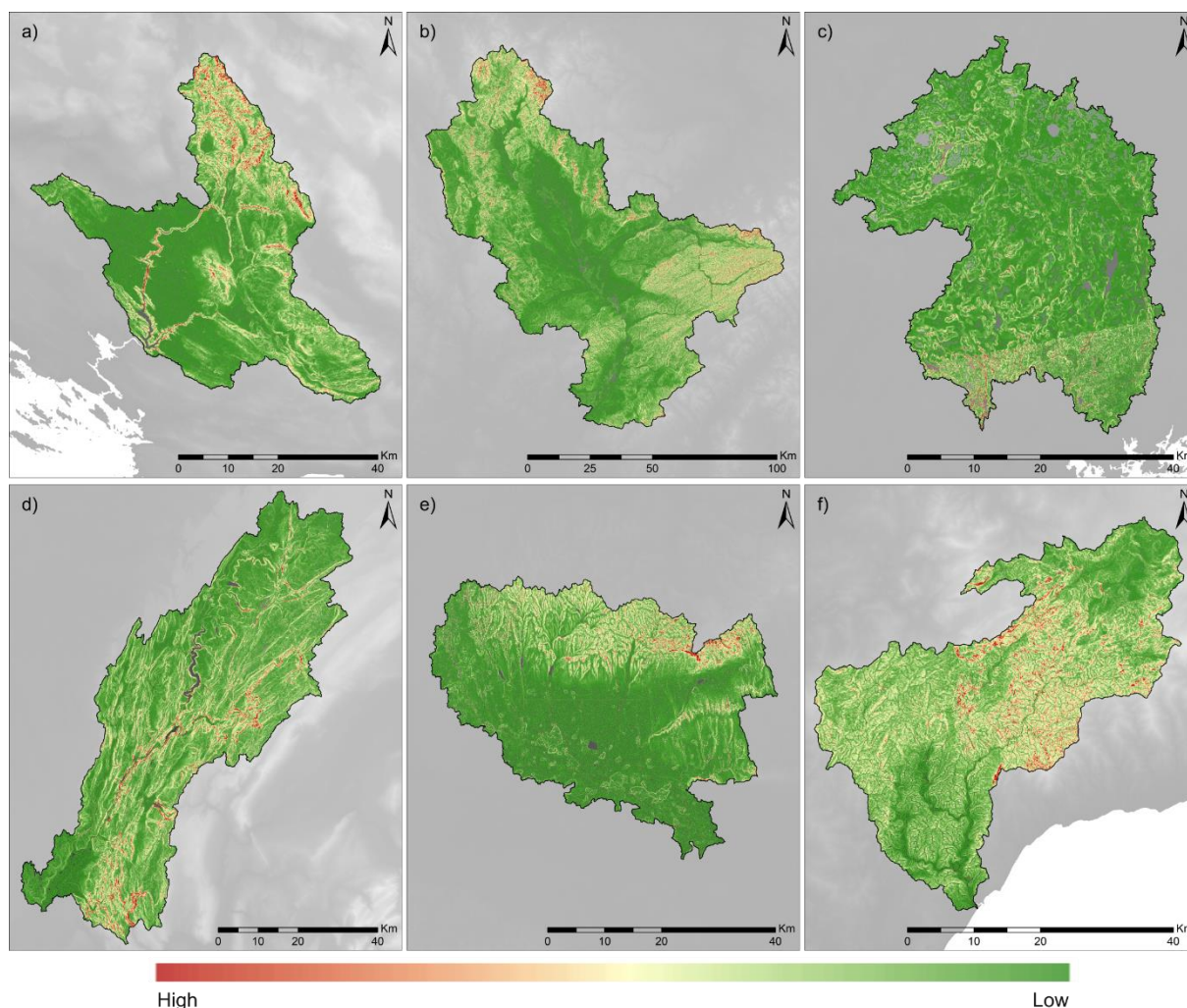


**Figure 13.** Water storage capacity in plains across DRNs: a) Croatia – Krka, b) Czech Republic – Morava, c) Finland – Vantaanjoki, d) France – Ain, e) Hungary – Fekete and f) Spain – Guadiaro.

## Erosion regulation

### Erosion regulation in slopes

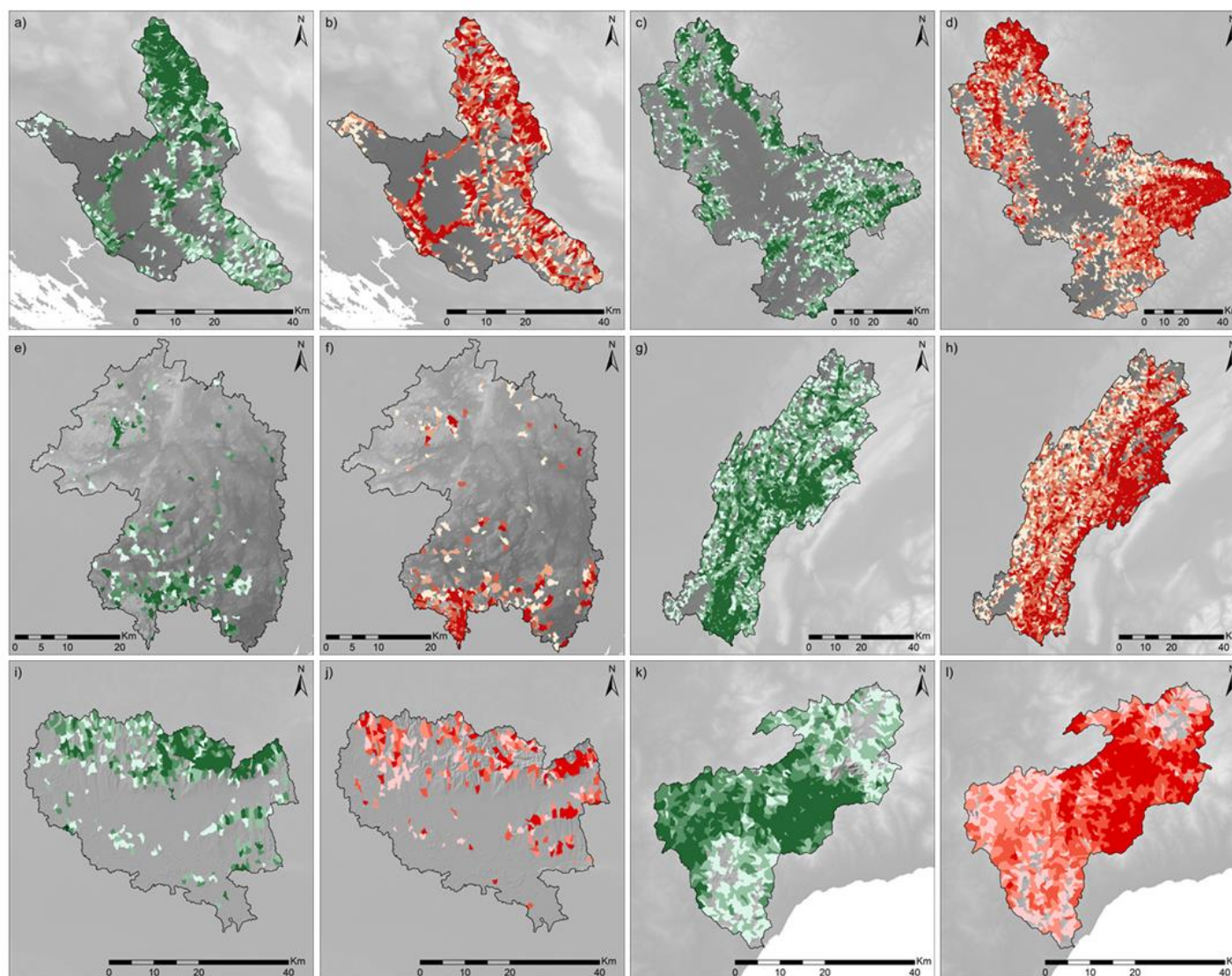
The spatial distribution of the Generic Erosion Potential (GEP), which was calculated to estimate the erosion regulation in slopes, showed a marked topographical pattern (Figure 14). Inside each study areas, pixels with higher GEP values occurred in places with steeper slopes, lower topographic convergence and larger local contributing areas.



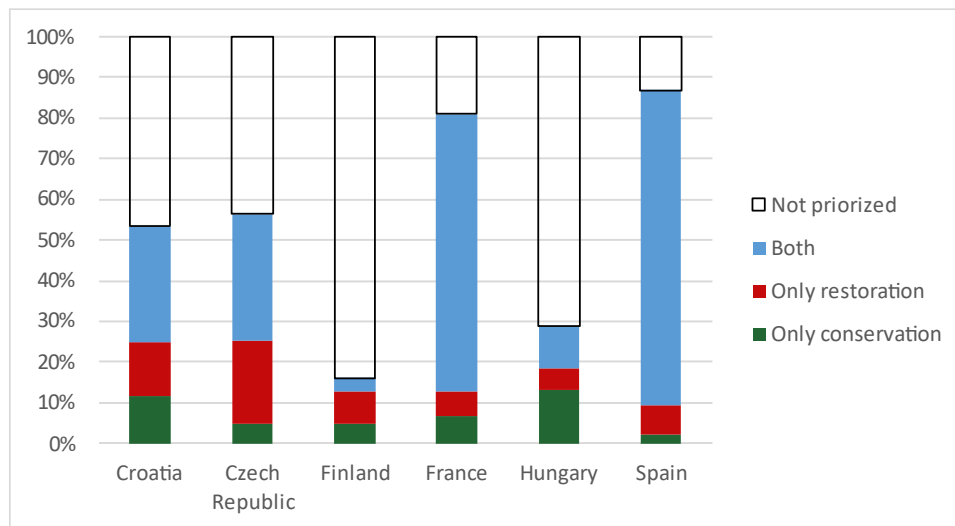
**Figure 14.** Generic Erosion Potential (GEP) for a) Croatia – Krka, b) Czech Republic – Morava, c) Finland – Vantaanjoki, d) France – Ain, e) Hungary – Fekete and f) Spain – Guadiaro.

After considering the presence of hillside forest, a prioritization of the drainage wings is proposed based on their potential to prevent sediments from being mobilized and arriving to the river network in each study area (Figure 15a, 15c, 15e, 15g, 15i and 15k). On the contrary, the lack of hillside forest allowed to prioritize those drainage wings based on their potential to contribute the most to erosion and delivery to the river network in each study area (Figure 15b, 15d, 15f, 15h, 15j and 15l). The proportion of prioritized wings in terms of erosion regulation in slopes considerably differed among cases of study (Figure 16). While in Guadiaro and Ain catchments most of the wings (around 80%) were prioritized for conservation, restoration or both, wings prioritized in Krka and Morava were around the half, being this proportion much lower for Vantaanjoki and Fekete. The predominance of drainage wings where it is proposed to conserve and restore at the same time is more noticeable in topographically heterogeneous areas with steeper slopes (such as in Guadiaro and Ain) suggesting that

under these environmental conditions forested areas play a crucial role as erosion regulation components.



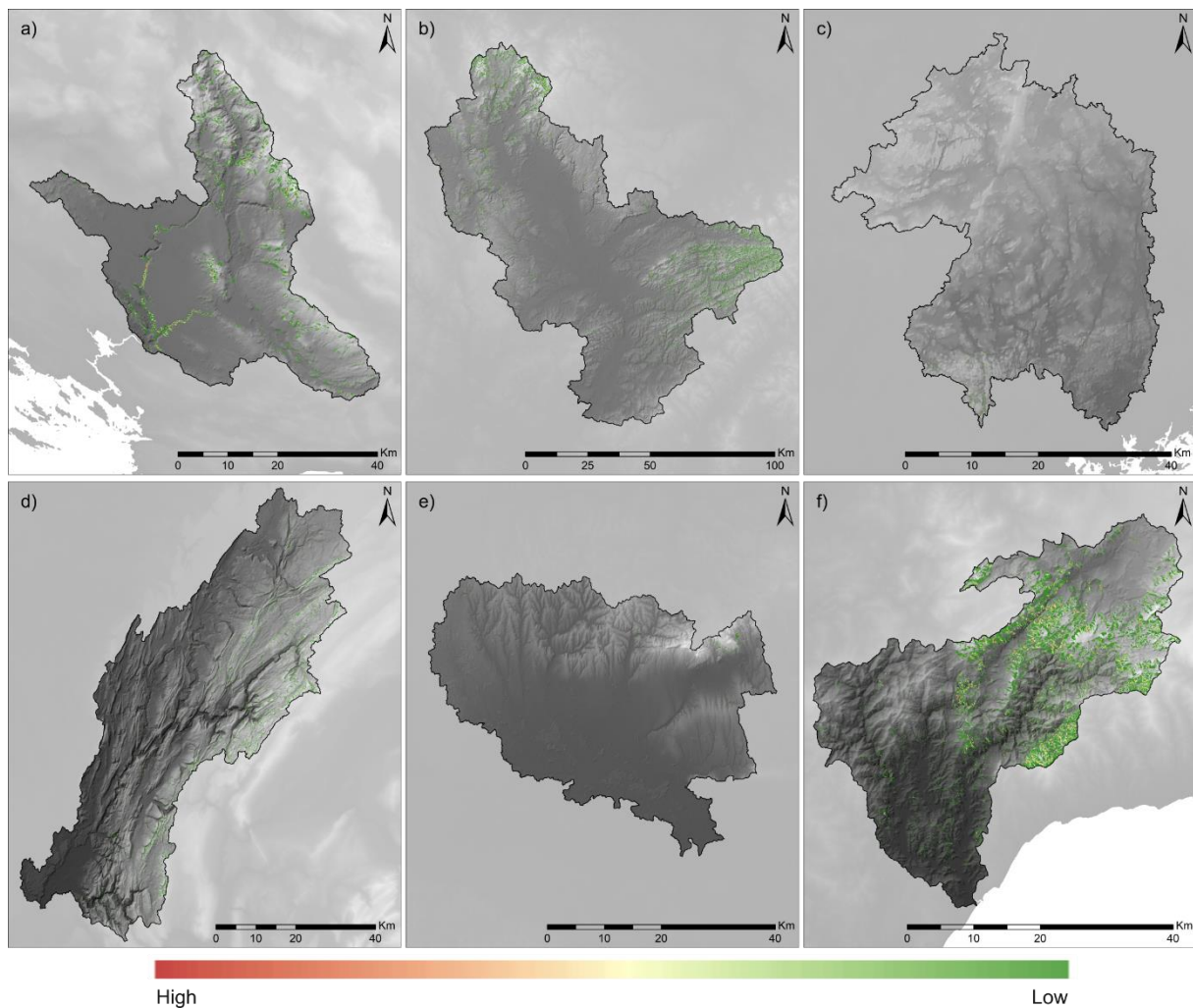
**Figure 15.** Proposed conservation (green) and restoration (red) drainage wings for the erosion regulation in slopes in a-b) Croatia – Krka , c-d) Czech Republic – Morava , e-f) Finland – Vantaanjoki , g-h) France – Ain , i-j) Hungary – Fekete , k-l) Spain – Guadiaro.



**Figure 16.** Proportion of prioritized drainage wings in terms of erosion regulation in slopes. DRNs: Croatia – Krka, Czech Republic – Morava, Finland – Vantaanjoki, France – Ain, Hungary – Fekete and Spain – Guadiaro.

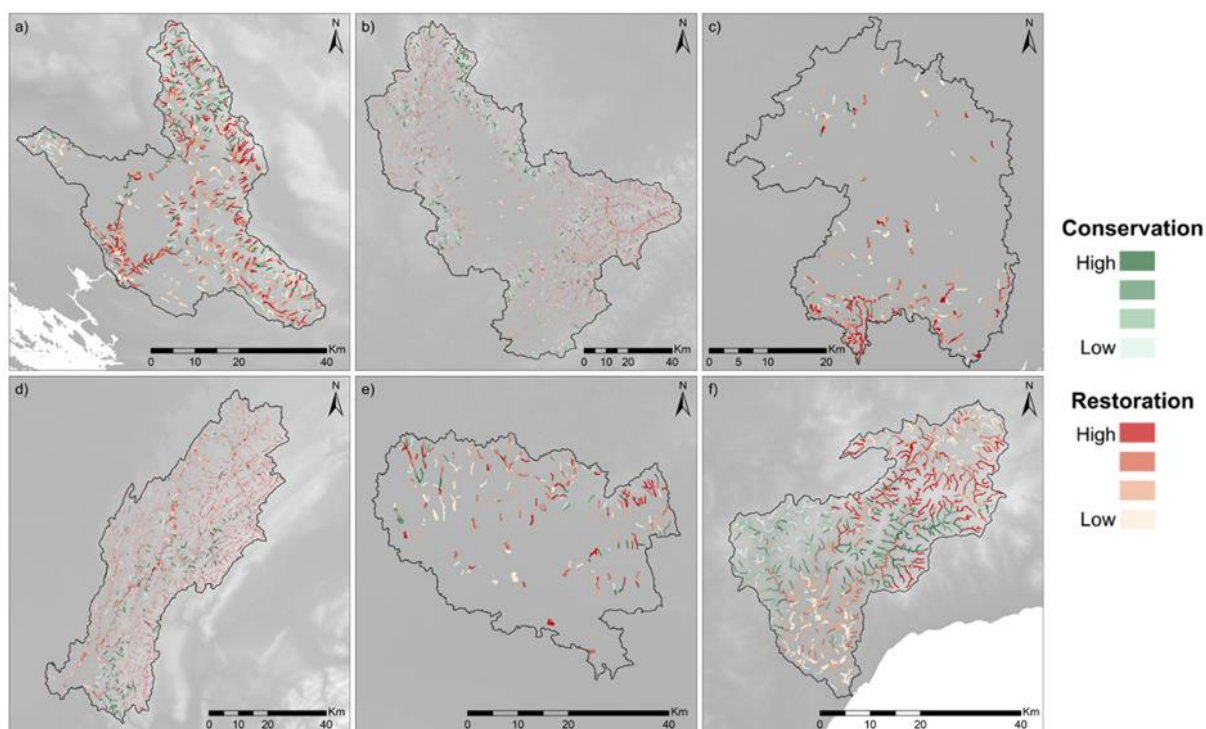
### Erosion regulation in riparian areas

The delivery of sediments in the river network derived from the absence of hillside forest in the drainage wings showed highly different patterns for the cases of study (Figure 17). In Fekete they are very scattered at the NE; in Vantaanjoki small areas concentrated at the mouth of the basin network; in Ain relative important areas are grouped in the east of the basin; in Krka larger areas with higher levels appear close to the mouth of the basin at the SE; in Morava two important isolated areas concentrate most of the delivery areas; and in Guadiaro a dense concentration of delivery values appears associated to mountainous areas in the mid-east of the basin.

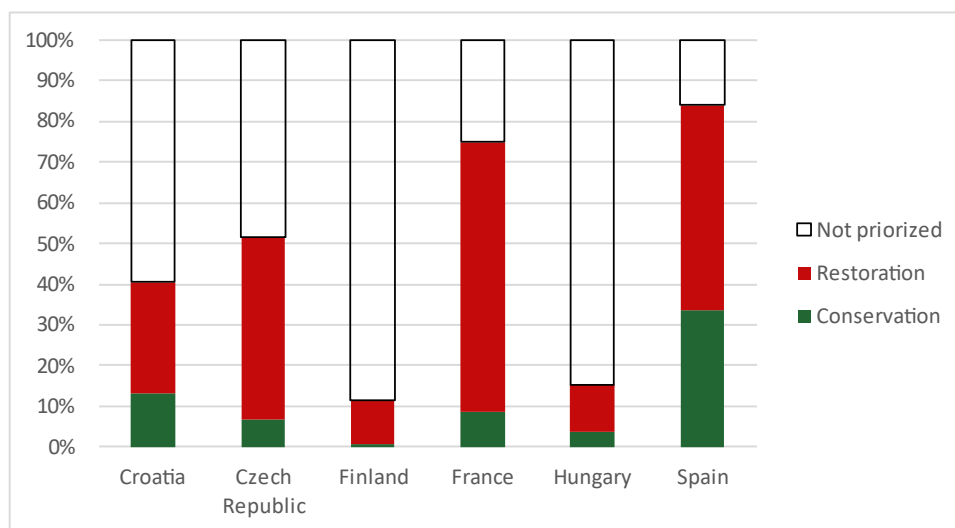


**Figure 17.** Generic Erosion Potential (GEP) potentially delivered to the reach from the drainage wing with absence of forest on hillside in a) Croatia – Krka, b) Czech Republic – Morava, c) Finland – Vantaanjoki, d) France – Ain, e) Hungary – Fekete and f) Spain – Guadiaro.

After considering the cover of riparian forest, a prioritization of the floodplain functional units is proposed based on their potential to prevent sediments from arriving to the river network by filtering in each study area or not (Figure 18). The proportion of prioritized floodplain functional units in terms of erosion regulation by riparian forests considerably differed among cases of study (Figure 19). While most of the floodplain functional units were prioritized for conservation or restoration in Guadiaro and Ain, in Krka and Morava were around the half, being this proportion much lower for Vantaanjoki and Fekete. In all cases the proportion of proposed UF for restoration was much higher than for conservation, as a consequence of the unbalanced cover of riparian forests in the reaches of floodplains across the basins. The low cover of riparian forested areas (<60%) in most of cases reduces the sediment filtering capacity and so, increases the amount of sediments delivered to the water bodies.



**Figure 18.** Proposed conservation (green) and restoration (red) floodplain functional units for the erosion regulation by riparian forests in a) Croatia – Krka, b) Czech Republic – Morava, c) Finland – Vantaanjoki, d) France – Ain, e) Hungary – Fekete and f) Spain – Guadiaro.



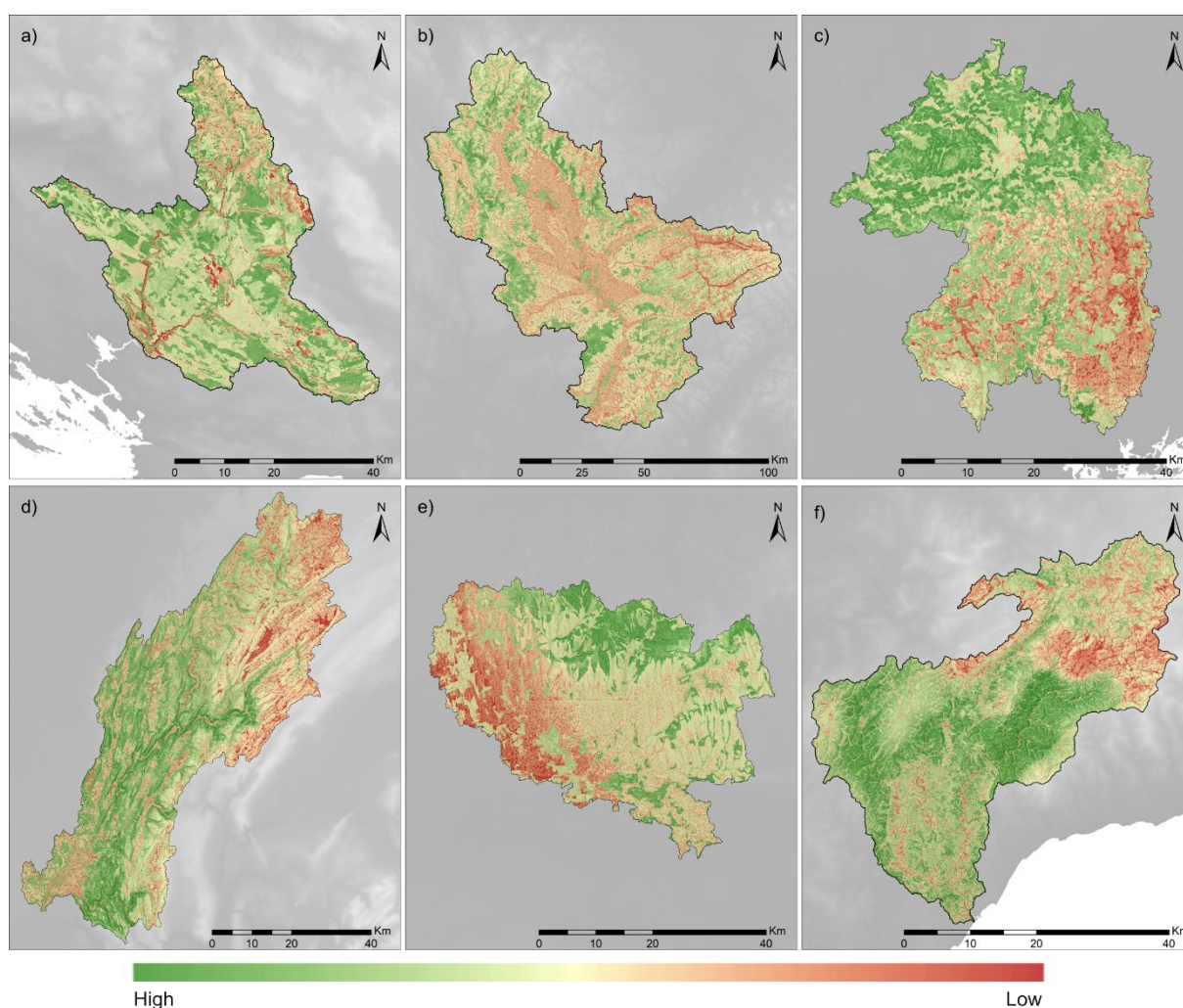
**Figure 19.** Proportion of prioritized floodplain functional units in terms of erosion regulation by riparian forests. DRNs: Croatia – Krka, Czech Republic – Morava, Finland – Vantaanjoki, France – Ain, Hungary – Fekete and Spain – Guadiaro.

## Drought regulation

### Surface Storage Index

The results show which areas within the catchment are the most suitable for holding water based in the biophysical components of the terrain (Figure 20), for each of the DRNs. In this case, higher values of the index mean more appropriate areas.

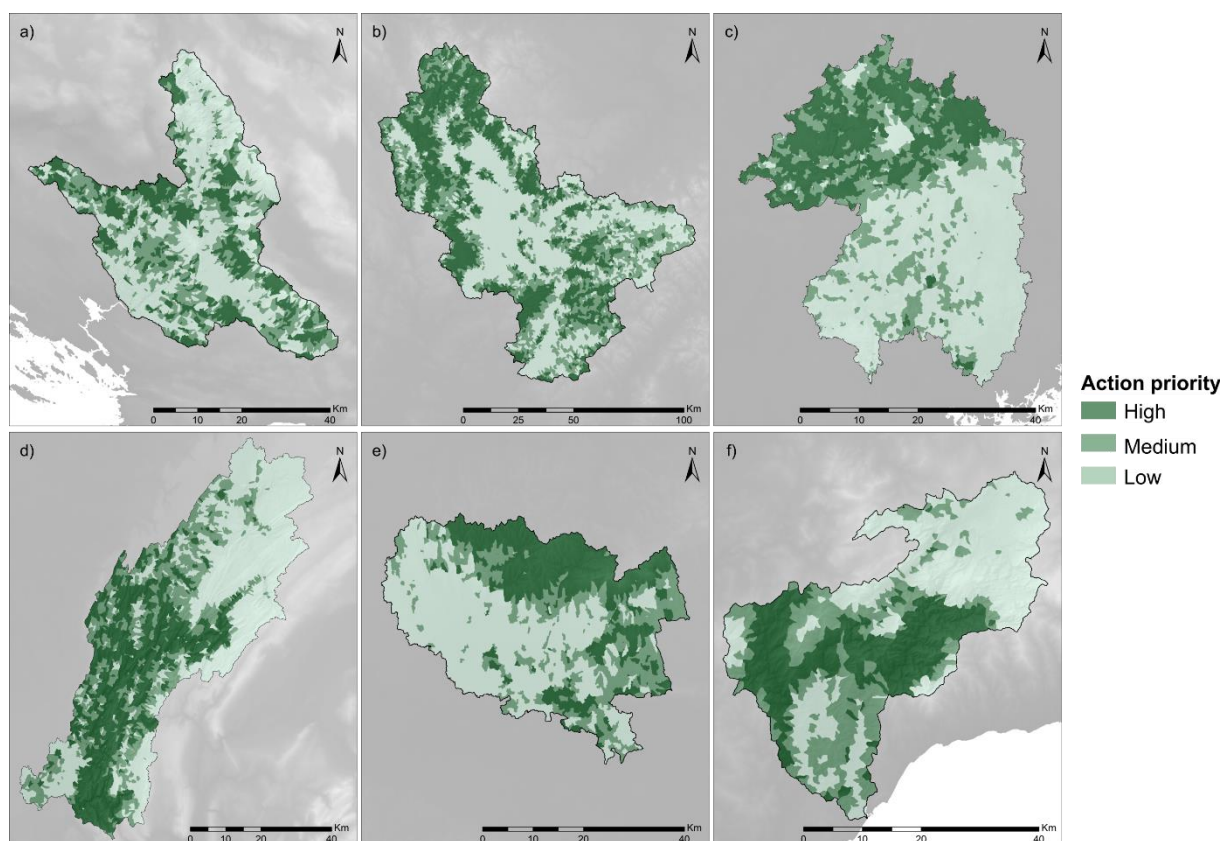
The study areas in Krka and the Morava (Figure 20a and b) do not show a homogeneous distribution in the mapping of areas suitable for surface storage. In Vantaanjoki the upper part of the catchment has the highest water retention potential (Figure 20c), while the eastern part in the middle and lower parts of the catchment concentrates the areas with the lowest potential. The Ain catchment has the highest values of the index in the middle zone, while the lowest values are found in the headwater areas (Figure 20d). The Fekete catchment has the most suitable areas for water storage mainly in the upper part of the catchment and the lowest values, less suitable areas, in the western part (Figure 20e). The Guadiaro catchment follows the same spatial pattern as the Ain, where we found the highest water storage capacity in the middle part of the catchment and the lowest capacity in headwaters areas (Figure 20f).



**Figure 20.** Surface Storage Index, related with the drought mitigation ecosystem service for a) Croatia – Krka, b) Czech Republic – Morava, c) Finland – Vantaanjoki, d) France – Ain, e) Hungary – Fekete and f) Spain – Guadiaro.

The amount of water that can be stored will depend on the amount of precipitation that reaches them. The steeper slope areas of watersheds tend to have a lower surface water storage potential because water moves quickly downslope, which also increases the risk of erosion and flooding downstream. In contrast, water tends to move more slowly in watershed areas with gentle slopes, which allows for storage. Areas with higher values of the TWI and convex plan curvatures (negative values) are prone to accumulate water. Vegetation covers, and forests specifically, have an important role in water retention as they can store large amounts of water in their biomass and soil, and slow down the movement of water through the catchment. The most favourable scenario for surface water storage was those areas with gentle slopes, convex terrain, high topographic humidity values and vegetation cover, mainly forests.

This index provides information on which parts of the land could provide the drought mitigation ecosystem service through surface water storage. Understanding the related factors can be very useful when designing and implementing nature-based solutions related to it, such as artificial wetlands, ponds, water gardens, among others. To this end, the information has been transferred to the functional work units, drainage wings, prioritizing them according to their potential (Figure 21). It is in the top class (75th percentile) on which the actions are primarily proposed to be carried out.

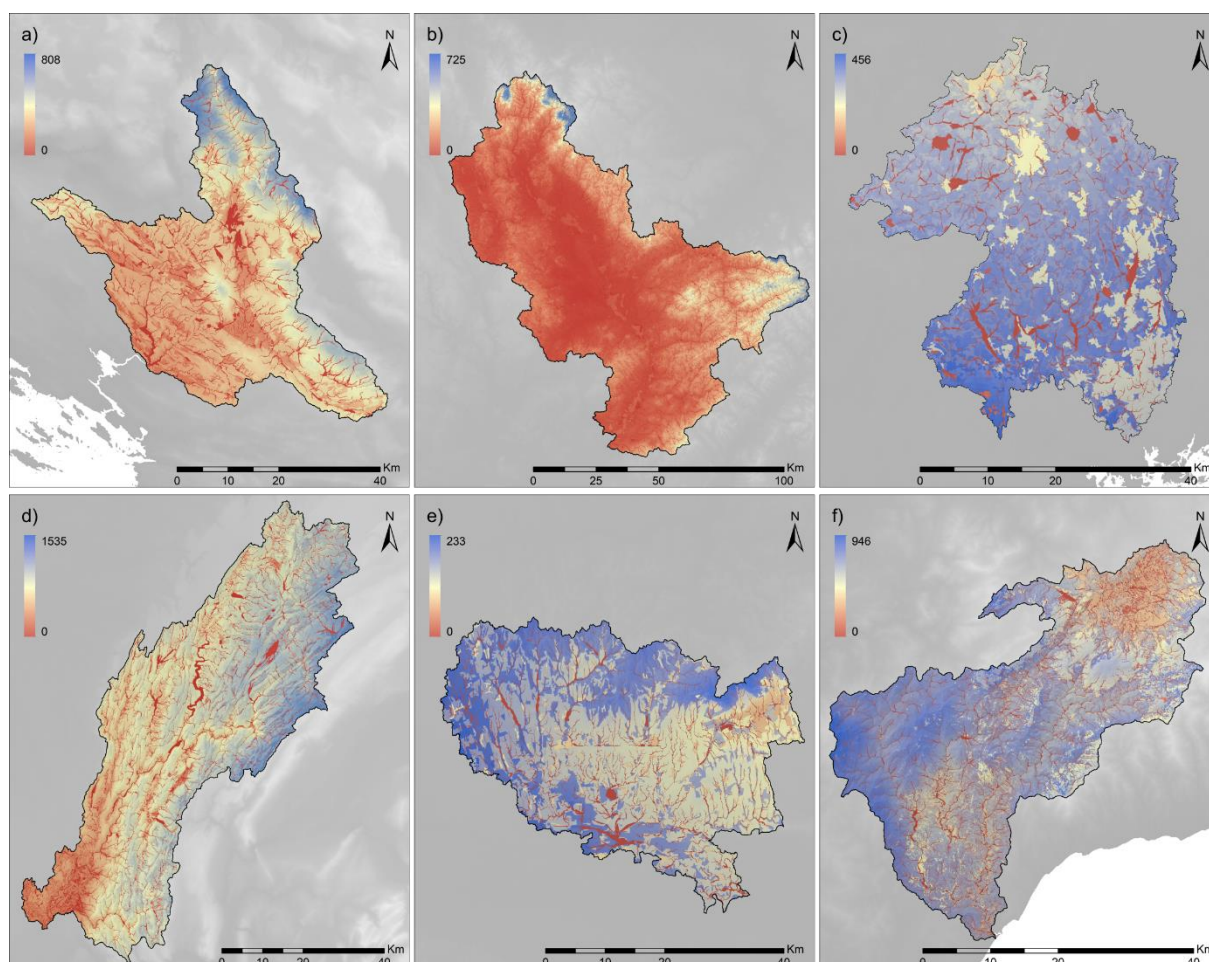


**Figure 21.** Proposed prioritization of drainage wings for drought mitigation through surface water storage in a) Croatia – Krka, b) Czech Republic – Morava, c) Finland – Vantaanjoki, d) France – Ain, e) Hungary – Fekete and f) Spain – Guadiaro.

Surface water storage act as a buffer between water supply and demand in the catchment during periods of drought, playing a crucial role in mitigating the impacts of the water shortage on the catchment. In addition, water is retained on the surface for a longer period, which increases the infiltration capacity of the area and, therefore, could also increase base flows. Water surface storage can also generate a lower hydrological response of a catchment by reducing the amount of water that flows directly into streams during rainfall events.

## Local Recharge Index

Local recharge index provides an estimation of the amount of water (in mm) that infiltrates into the soil over a year and could potentially recharge the groundwater (Figure 22). Higher values of the index indicate higher infiltration, where the magnitude of the value is defined by the amount of precipitation. The index's strength lies in its capacity to assess the water yield spatially, showing the groundwater recharge areas, thus it can be used to identify the location of areas with high or low rates.



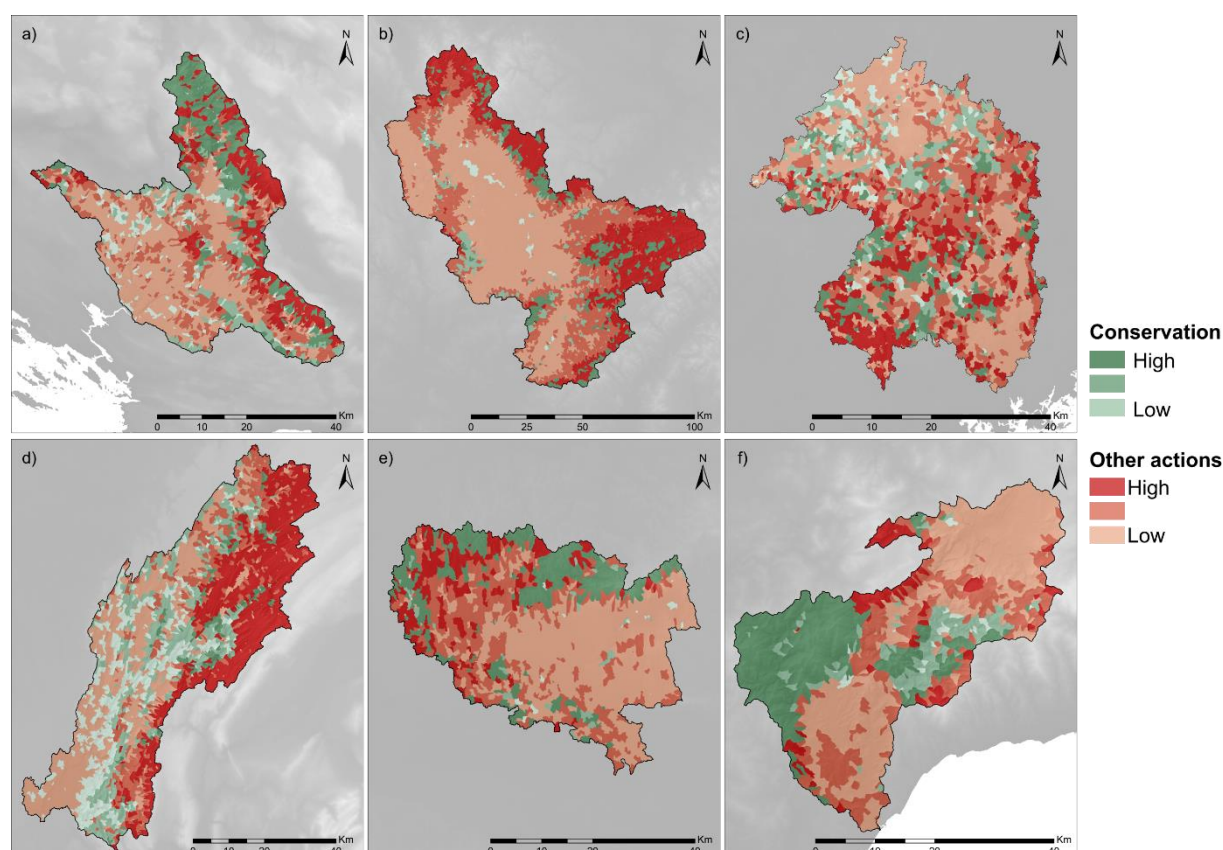
**Figure 22.** Local Recharge Index (in mm), related with the drought mitigation ecosystem service for a) Croatia – Krka, b) Czech Republic – Morava, c) Finland – Vantaanjoki, d) France – Ain, e) Hungary – Fekete and f) Spain – Guadiaro.

The results show differences between catchments in the spatial distribution of areas with high recharge rates. The Krka, Ain and especially the Morava catchments (Figures 22a, 22b, 22d) are those where there are fewer and moreover very localised areas of high infiltration. Vantaanjoki, Fekete and Guadiaro (Figures 22c, 22e, 22f) are the cases where high infiltration values are more distributed over the whole catchment.

Higher values of the LRI index are found, for all the case studies, where precipitation is the highest within the catchments. Lower values are related to higher surface runoff generation and to areas where evapotranspiration rates are higher, mainly found in agricultural lands. In this sense, changes in land use and land cover may have a significant impact on water resources. For example, areas with

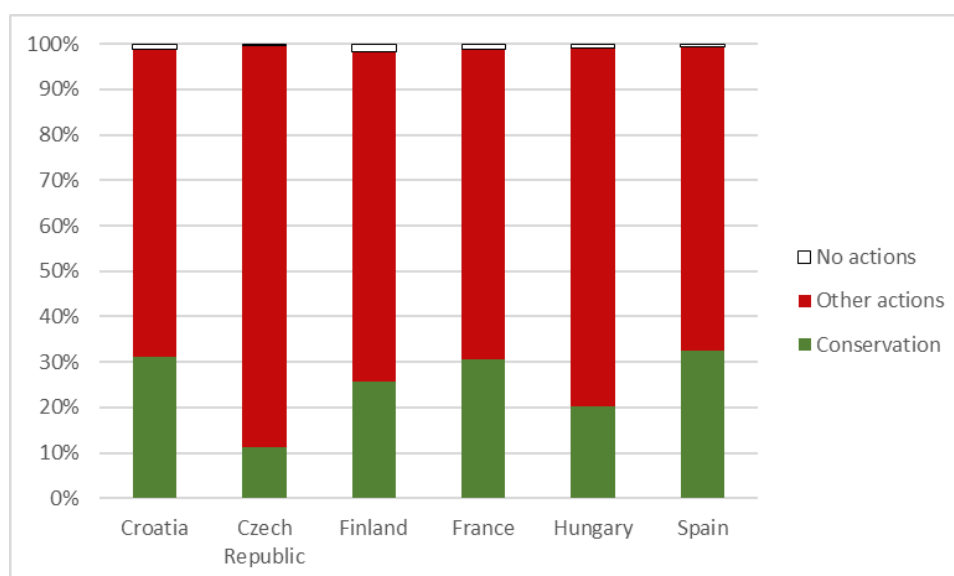
low rates of local recharge may be more sensitive to changes that reduce evapotranspiration and increase local recharge.

This index can be used to identify areas where groundwater recharge is likely to be high, and where additional management strategies may be required to protect and enhance groundwater resources. In this sense, drainage wings with more than 60% of forest are those where the ecosystem service is guaranteed, and are proposed as areas to be conserved, while those where there is no such land coverage, may require additional measures to promote local recharge, to increase the amount of water stored in the soil and groundwater (Figure 23). Strategies that focus on increasing local recharge, such as promoting the use of nature-based solutions, can help to improve water security and reduce the impacts of drought on natural systems.



**Figure 23.** Proposed conservation (green) and other actions (red) drainage wings to generate local recharge in a) Croatia – Krka, b) Czech Republic – Morava, c) Finland – Vantaanjoki, d) France – Ain, e) Hungary – Fekete and f) Spain – Guadiaro.

The proportion of functional units for each type of action proposed are shown in Figure 24. The main. It can be observed that specific management and planning activities need to be carried out for all catchments, as the proportion of drainage wings where conservation actions need to be carried out is below 33% in all catchments. This is specifically relevant for the Morava catchment, where only 11% of the territory have the ES guaranteed.

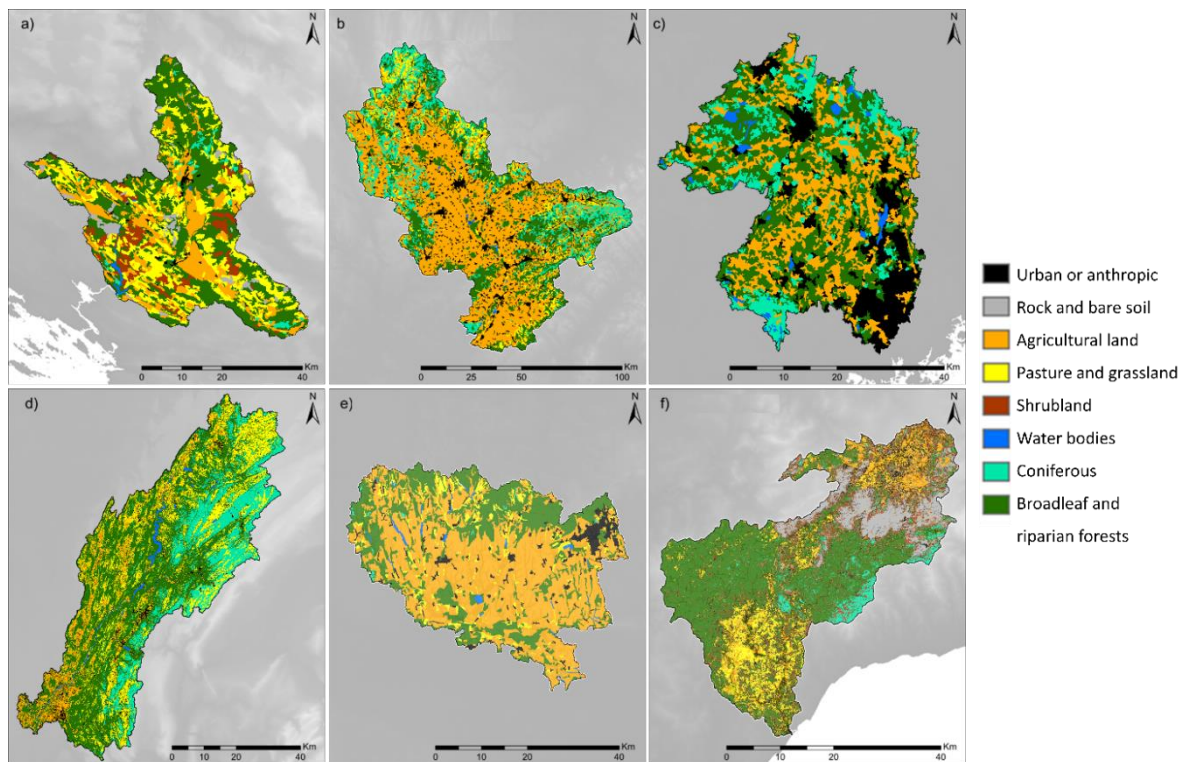


**Figure 24.** Proportion of prioritized drainage wings in terms of local recharge capacity. DRNs: Croatia – Krka, Czech Republic – Morava, Finland – Vantaanjoki, France – Ain, Hungary – Fekete and Spain – Guadiaro.

## Thermal regulation

The spatial distribution of thermal regulation across DRNs shows a clear dependence of LULC patterns, i.e. shadow cast is directly related to the occurrence of forest cover (see Figure 25 below), and a recognized morphodynamic control. Across basins, pixels with higher thermal reductions occur in larger valley bottoms with steeper slopes and larger local contributing areas.

The more widespread distribution of forests can indeed have a positive impact on reducing water temperature in European rivers (Feld et al., 2011). Forests play a crucial role in maintaining the health and balance of ecosystems, including river functioning and related services (Morri et al., 2014). Tree canopy provides shade, which helps to block direct sunlight from reaching the water's surface, prevents excessive heating and helps maintain cooler temperatures. Trees also absorb water from the ground through their roots and release it into the atmosphere through evapotranspiration. This moisture-laden air cools down the surrounding environment, including the river, through evaporative cooling. Riparian buffers also play a role at this regard across entire valley bottoms, creating a buffer zone between land fluxes and streams that helps to regulate water temperature by providing shade and acting as a natural barrier to block excess heat from adjacent land areas. They also contribute filtering rainwater and reducing surface runoff. This process helps maintain overall water quality parameters in rivers by reducing the influx of sediments and pollutants, which can contribute to increased water temperatures by initializing metabolic cascades. Finally, tree roots help stabilizing the river banks and the entire ecosystem functioning across alluvial terraces, reducing erosion and maintaining the natural water flows (superficial and subsuperficial). This stabilization prevents excessive sedimentation, which can increase water temperature by trapping heat and impeding natural cooling processes.



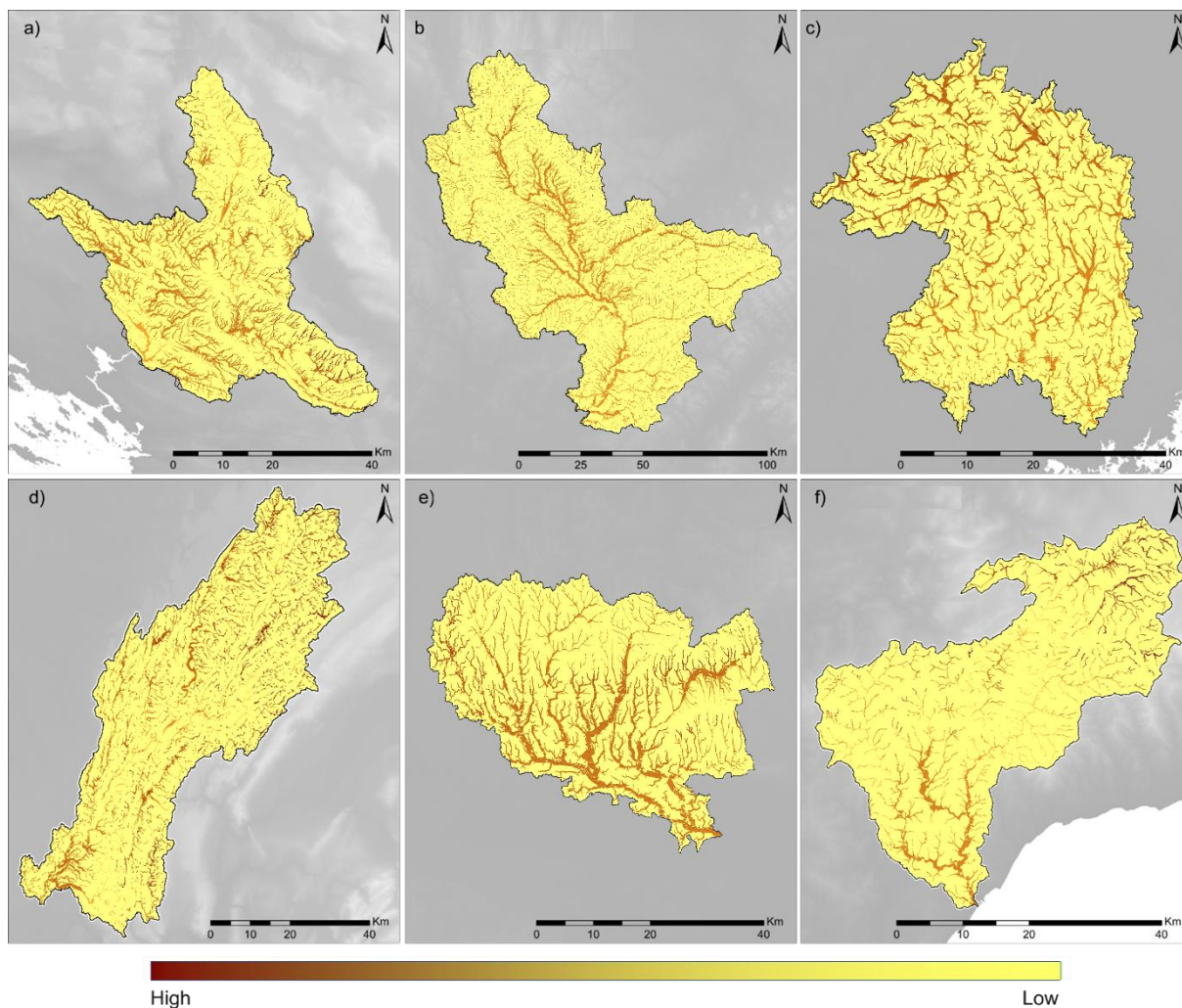
**Figure 25.** Land use and land cover distribution across DRNs: a) Croatia – Krka, b) Czech Republic – Morava, c) Finland – Vantaanjoki, d) France – Ain, e) Hungary – Fekete and f) Spain – Guadiaro. Broadleaf forest category includes riparian forests occupying valley bottoms.

While preserving and restoring forests can help reducing water temperature, contributing significantly to maintain healthier and cooler river ecosystems, they are not the solely factor influencing it. Other remarkable factor contributing notably to buffering stream and river temperature are groundwater discharge areas such as spring sources, streambed resurgences or seepages. However, accounting for the effect of this other factor across the 6 case studies is beyond the scope of this study.

After identifying the presence of riparian forests across entire valley bottoms, total radiation was defined for each DRN based as their total insolation received during summer months (Figure 26). The amount and distribution of solar radiation in the catchments (i.e. in the areas identified as valley bottoms) varies based on the topography and geographical location (Dobrowski, 2011). In flat basins, solar radiation is typically evenly distributed across the entire area, assuming there are no geomorphological obstructions for the amount of insolation received in valley bottoms and streams. The intensity of solar radiation may vary slightly depending on local atmospheric conditions, but there are no significant deviations caused by topography alone. Basins with steep-sided topography, typical of higher altitudes, usually present deep valleys and can experience larger variations in solar radiation distribution. The slopes of the basin can cause shading, especially on the north-facing slopes, where direct sunlight may be obstructed by the terrain features. South-facing slopes receive more solar radiation throughout the day due to their inclination towards the sun. Convex-shaped basins, where the surrounding terrain rises towards the basin's center, tend to accumulate more solar radiation. The shape of the basin can act as a natural amphitheater, capturing and concentrating sunlight. As a result, the base of a convex valley may receive higher solar radiation compared to the surrounding areas. Oppositely, concave-shaped basins, where the surrounding terrain slopes inward towards the center, can experience lower solar radiation levels in the base. The shape of the basin can cause shading and

obstruction of direct sunlight. The reduced solar radiation can result in cooler temperatures and different vegetation patterns in the basin.

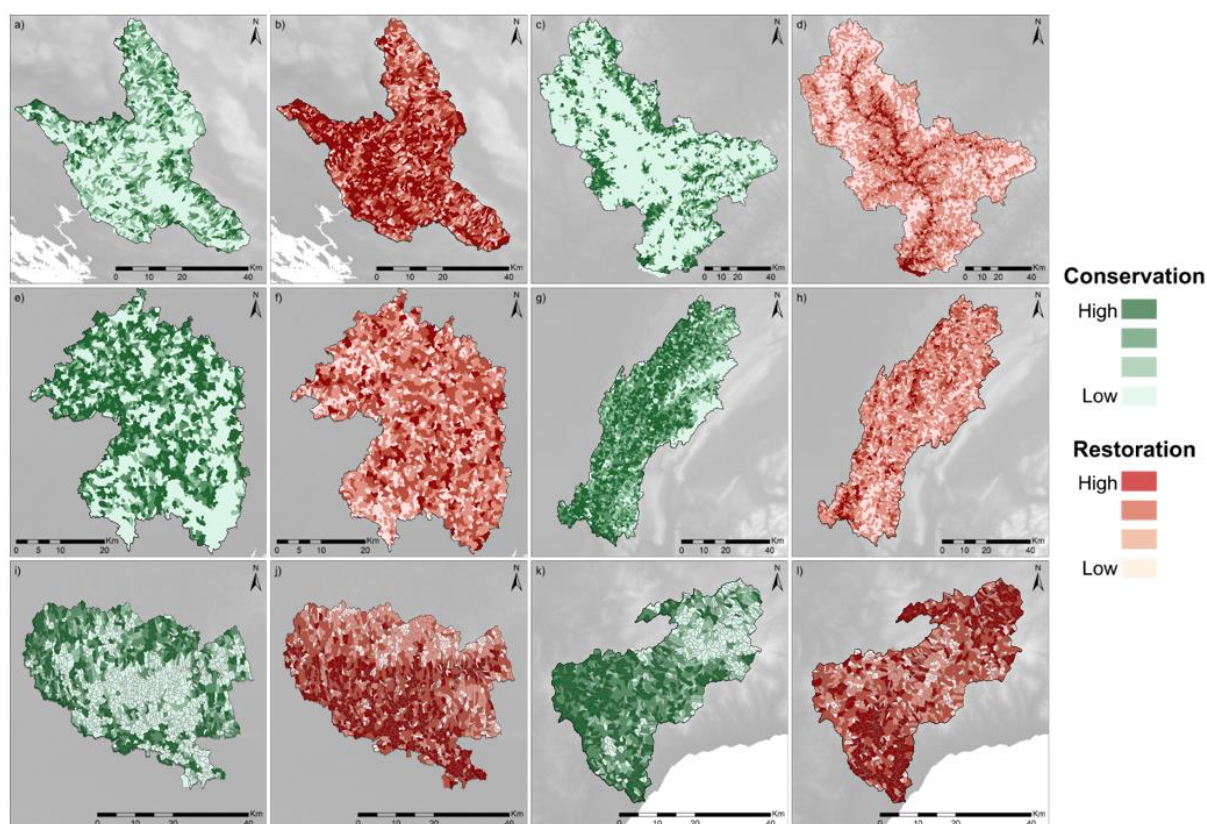
In addition to geomorphology, there are specific local factors such as latitude, elevation, climate and mainly the presence of vegetation, as explained before, that can further influence the distribution of solar radiation in a basin. Additionally, seasonal variations and weather patterns will also affect the solar radiation received in different topographic areas of the basin (Paruelo et al., 2005).



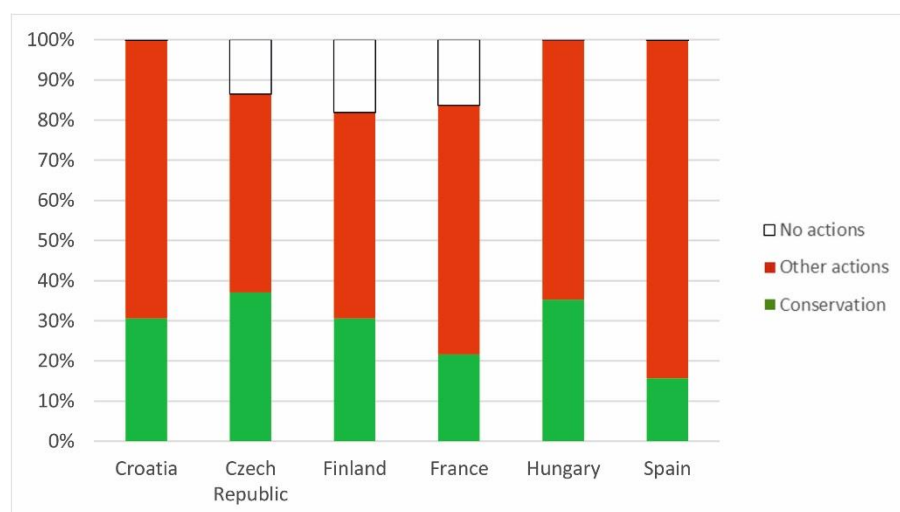
**Figure 26.** Incoming solar radiation or insolation received by all DRN valleys independently of having or not riparian forest coverage: a) Croatia – Krka, b) Czech Republic – Morava, c) Finland – Vantaanjoki, d) France – Ain, e) Hungary – Fekete and f) Spain – Guadiaro. The total amount of radiation calculated for a particular location correspond to all direct, diffuse and global insolation.

In the base of maximum potential insolation received by valley floors and the amount and distribution of vegetation cover, casting shadow to main streams and other functional areas of valley bottoms related to water temperature regulation, we identified and prioritized drainage wings based on their potential to contribute the most to thermal reduction across each study area (Figure 27). The proportion of prioritized wings in terms of thermal regulation in slopes considerably differed among cases of study. In terms of territory management for this ES, while in the Guadiaro, Fekete and Krka catchments all wings are identified as candidates for restoration (or conservation) actions, in the Ain, Vantaanjoki and Morava from 15 to 20% are never selected. Nevertheless, all share a common pattern of having no functional units with conservation requirements only (Figure 28). This is related to a double control, i.e. the heterogeneous topography and the varying amount of riparian forests across valleys in all DRNs. In turn, there is a large amount of drainage wings on which it is proposed to

conserve and restore land at the same time. At this regard, as an example, conservation of the Fekete basin shows the existence of no functional units selected across the entire center of the basin, because of the total absence of riparian forests (see Figure 26 above), while in the south of Guadairo basin both restoration and conservation actions are overlapped. They correspond to bigger valleys with fragmented patterns of riparian forests.



**Figure 27.** Proposed conservation (green) and restoration (red) drainage wings for the erosion regulation by riparian forests in a-b) Croatia – Krka , c-d) Czech Republic – Morava , e-f) Finland – Vantaanjoki , g-h) France – Ain , i-j) Hungary – Fekete , k-l) Spain – Guadiaro.



**Figure 28.** Proportion of prioritized functional units in terms of thermal regulation by riparian forests. DRNs: Croatia – Krka, Czech Republic – Morava, Finland – Vantaanjoki, France – Ain, Hungary – Fekete and Spain – Guadiaro.

## Carbon emissions

With the predicted CO<sub>2</sub> flux rates (mg Cm<sup>-2</sup>d<sup>-1</sup>) and the calculated daily flow and dry surface for each river reach, we calculated for each river segment the daily total CO<sub>2</sub> emissions. Then, we calculated the acumulative carbon from CO<sub>2</sub> (C-CO<sub>2</sub>) emissions for each of the selected periods, split in emissions from flowing waters and dry riverbed. The contribution of the emissions from riverbeds and flowing water for each period and DRN is summarized in Table 8. We then classified for each DRN the emissions at the reach scale in 3 categories: low (percentile 0 to 0.33), medium (percentile 0.33 to 0.66) and high emissions (percentile 0.66 to 01) (Figures 29 to 34).

**Table 8.** Cotribution of emissions from flowing water and dry riverbed to total C-CO<sub>2</sub> emissions for each DRN (total of all river reaches) for each of the selected periods. Pre-dry period: march-april; Dry period: july-august; Post-dry period: november-december

	Pre-dry phase		Dry phase		Post-dry phase	
	prop. C.flow	prop. C.dry	prop. C.flow	prop. C.dry	prop. C.flow	prop. C.dry
<b>Ain-Albarine</b>	0.79	0.21	0.66	0.34	0.89	0.11
<b>Vantaanjoki - Lepsämäenjoki</b>	0.93	0.07	0.67	0.33	0.92	0.08
<b>Krka-Butižnica</b>	0.86	0.14	0.53	0.47	0.94	0.06
<b>Fekete-Bükkösdi</b>	0.40	0.6	0.12	0.88	0.28	0.72
<b>Guadiaro-Genal</b>	0.56	0.44	0.15	0.85	0.58	0.42
<b>Morava-Velička</b>	0.60	0.4	0.10	0.90	0.54	0.46

C-CO<sub>2</sub> emissions varied between river basins, with Krka-Butižnica (Figure 29) having the lowest evasion per segment and the Morava-Velička (Figure 30) the highest. Additionally, emissions also varied between flow phases although the magnitude of that changes varied between DRNs. Results show three different C-CO<sub>2</sub> emission regimes. For example, in Vantaanjoki-Lepsämäenjoki (Figure 31) and Guadiaro-Genal (Figure 32) emissions decreased during the dry phase, while in Morava-Velička (Figure 30), Ain-Albarine (Figure 33) and Fekete-Bükkösdi (Figure 34) they increased during that same period. Finally, in Krka-Butižnica emissions gradually decreased from the Pre-dry to the Post-dry phase (Figure 29).

These C-CO<sub>2</sub> evasion patterns do not seem to be linearly related to drought as emissions increase during the dry phase in both Vantaanjoki-Lepsämäenjoki, which has the lowest dry sediment contribution (33 %) and Genal, which has one of the highest dry sediment contribution (85%, Table 8). In the two DRNs with the highest dry sediment contribution to CO<sub>2</sub> emissions during the dry period (Fekete-Bükkösdi and Morava-Velička) the total carbon evasion increased during that phase. That could indicate that lower flows and a higher proportion of dry streams boost C-CO<sub>2</sub> evasion from rivers. A likely mechanism for this emission increase could be the low resistance of autotrophic organisms and the higher resistance of heterotrophs to desiccation (Sabater et al., 2016; Timoner et al., 2012). However, this pattern is not shared by all six DRNs, meaning some other environmental condition may be controlling carbon emissions. For example, the Genal DRN may present very low sediment moisture through the Dry phase due to sustained high temperatures (> 30 °C) and rain absence, which can hinder dry sediment respiration, thus reducing total CO<sub>2</sub> emissions (Sabater et al., 2016). After rewetting, the

proportion of CO<sub>2</sub> emissions from dry and wetted areas are similar to those found in the Pre-dry phase in five out of the six DRNs. This could indicate the main processes controlling C-CO<sub>2</sub> emissions during flowing conditions were recovered to Pre-dry levels.

The two main sources of inorganic C emissions are inorganic carbon originating in terrestrial ecosystems and carried underground to rivers and River Ecosystem Metabolism (REM) (Hotchkiss et al., 2015). Concerning inorganic carbon emissions from terrestrial sources, which are thought to be the primary source of CO<sub>2</sub> evasion from rivers, subterranean flow regulates CO<sub>2</sub> transit from terrestrial ecosystems to rivers. This movement may be reduced or even interrupted during low flows or total flow cessation (Dry phase), limiting its contribution to river CO<sub>2</sub> emissions. Moreover, terrestrial soil respiration also follows a seasonal pattern with biological activity being regulated by light, temperature and precipitation regimes, which could have an effect on river emission patterns through the year (Bernhardt et al., 2018).

On the other hand, REM is estimated to contribute up to 28% of CO<sub>2</sub> emissions from rivers (Hotchkiss et al., 2015). This percentage would vary along the river network as well as seasonal GPP and ER vary greatly with light, temperature and discharge regimes. GPP is usually higher during spring and summer due to higher temperatures and light availability, contributing to carbon sequestration while ER is more stable through the year and higher than GPP due to allochthonous organic matter inputs (Battin et al., 2023). In the future, we will add REM estimates to C-CO<sub>2</sub> emission in DRYvER DRNs to increase our understanding of metabolism's contribution to carbon evasion from drying rivers, which account for up to 60% of the world's water courses (Messenger et al., 2021). Higher GPP rates, for example, may not contribute significantly to lower total carbon emissions at the catchment level if the high primary production season overlaps with the Dry phase, which is characterized by partial or total channel desiccation.

As drying streams appear to boost C-CO<sub>2</sub> evasion, contribution of river networks to global carbon emissions could be severely underestimated (Marcé et al., 2019). Thus, an in-depth assessment of global carbon emissions of non-perennial rivers should be prioritized. The effect of periodic dry periods on C-CO<sub>2</sub> evasion should be carefully analyzed, especially as under future climate change conditions their frequency and severity are expected to increase. Increasing drought severity could increase global carbon emissions, creating a positive feedback loop that could further contribute to the self-perpetuation and intensification of climate change (Armstrong McKay et al., 2022). This CO<sub>2</sub> emission model could be applied to future climate change scenarios to project the effect of a change on climate in carbon emissions from rivers. Further understanding of river C-CO<sub>2</sub> sources, their drivers and spatio-temporal dynamics could aid in catchment management for avoiding the most dangerous global change scenarios. Nature-based solutions and integrated catchment management of land uses and other anthropogenic infrastructures and activities (hydrological barriers, sewage effluents, water abstraction, etc.) could help regulate river carbon emissions and restore damaged freshwater ecosystem functions and services.

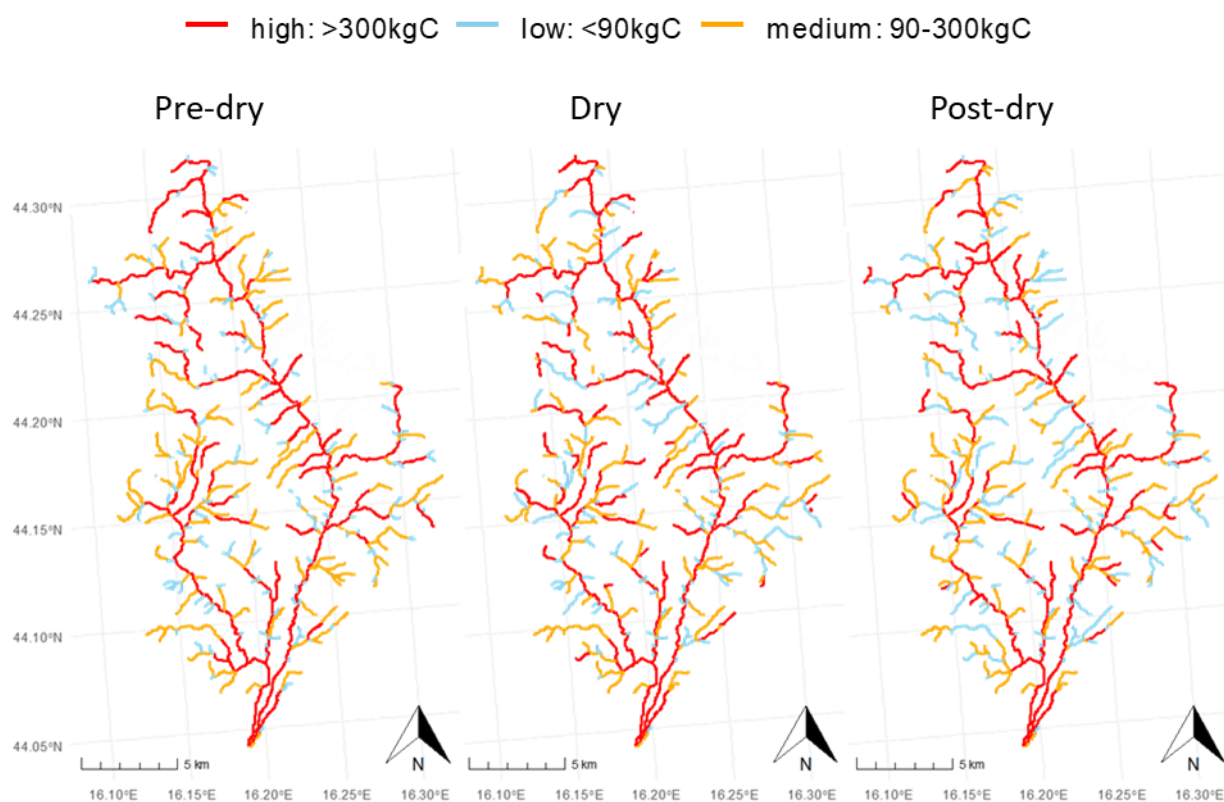


Figure 29. Total Krka-Butižnica DRN C-CO<sub>2</sub> emissions by river segment during each of the flowing phases.

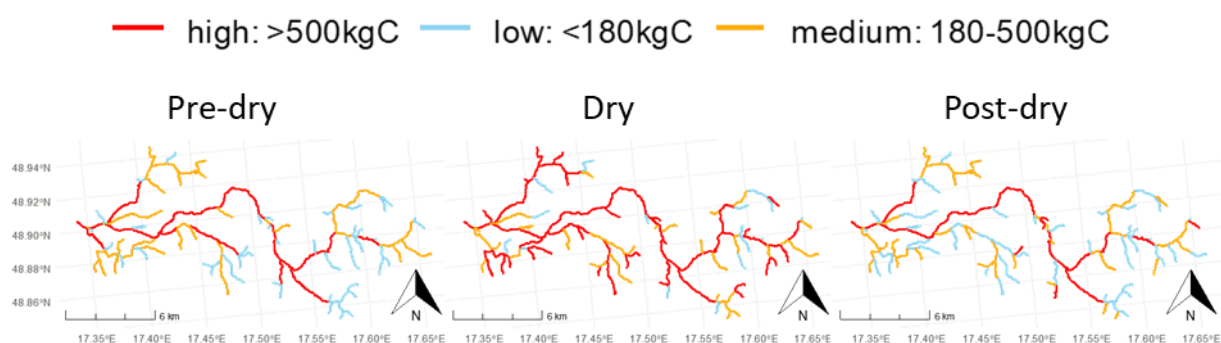


Figure 30. Total Morava-Velička DRN C-CO<sub>2</sub> emissions by river segment during each of the flowing phases.

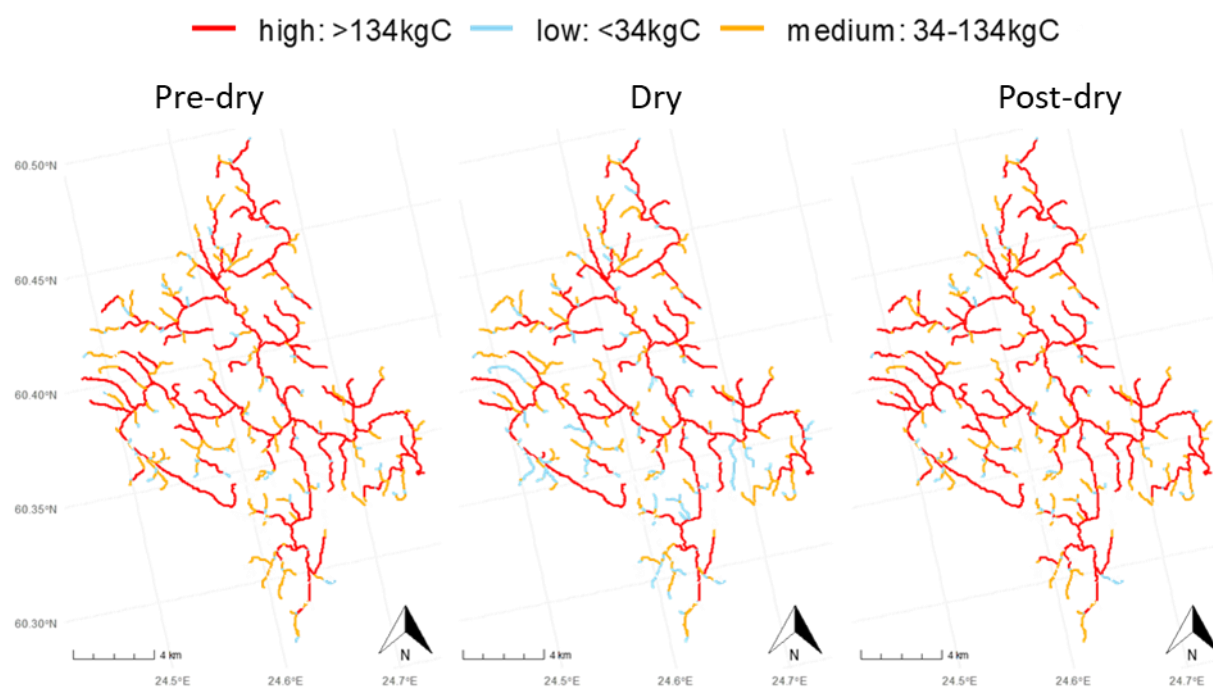


Figure 31. Total Vantaanjoki-Lepsämäenjoki DRN C-CO<sub>2</sub> emissions by river segment during each of the flowing phases.

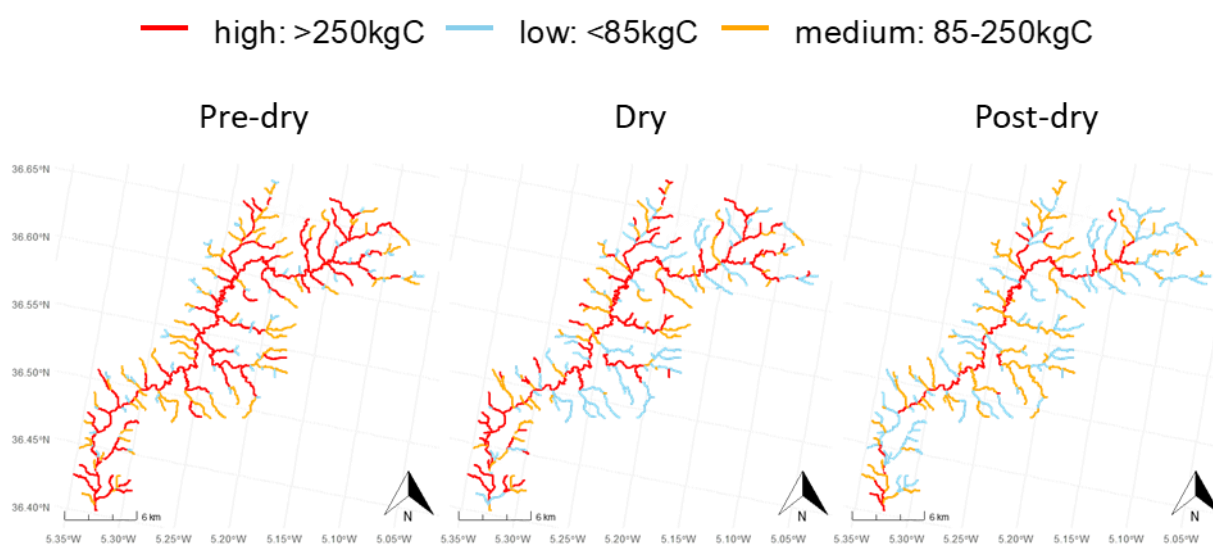


Figure 32. Total Guadiaro-Genal DRN C-CO<sub>2</sub> emissions by river segment during each of the flowing phases.

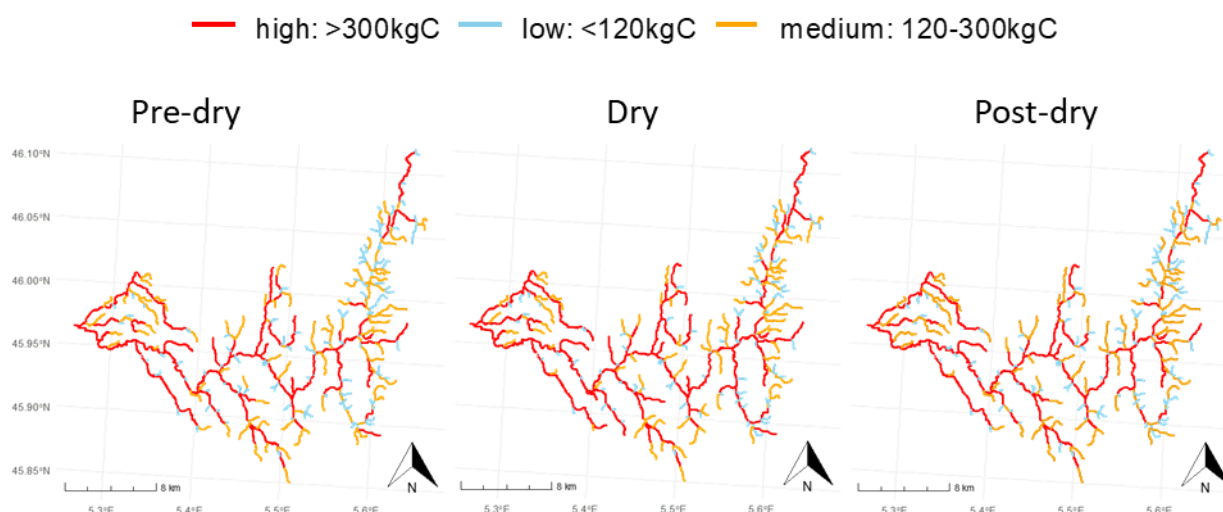


Figure 33. Total Ain-Albarine DRN C-CO<sub>2</sub> emissions by river segment during each of the flowing phases.

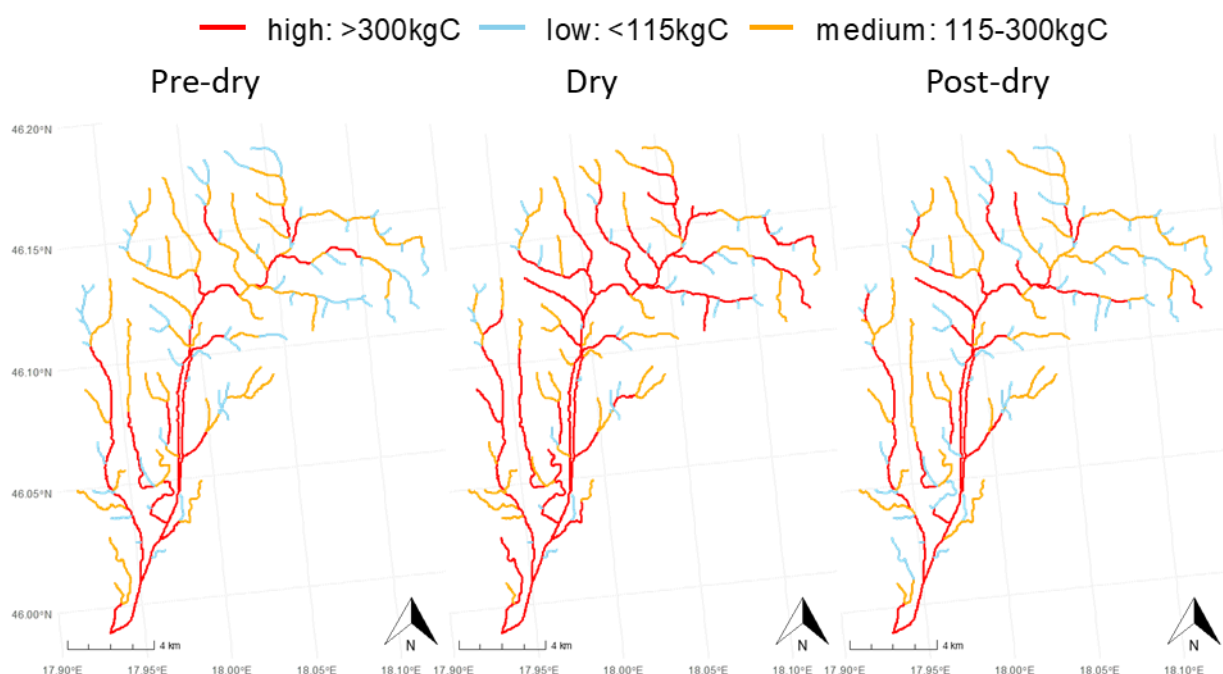


Figure 34. Total Fekete-Bükkösi DRN C-CO<sub>2</sub> emissions by river segment during each of the flowing phases.

## General discussion for ES provision in DRNs

### ES provision and intermittence patterns of DRNs

Drying river networks (DRNs) refer to river systems that experience temporary or permanent reduction in water flow due to factors related to phenological climatic conditions exacerbated by climate change events (Gudmundsson et al., 2021; Shanafield et al., 2021). This drying of river networks has significant impacts on the provision of ecosystem services (ES) and its magnitude and affects depend largely on the spatial configuration of landscape patterns across both basins and river networks. One step beyond, the relationship between ES provision and intermittence patterns in DRNs is complex and can

vary depending on the specific context. In some cases, the reduction in water flow may lead to a decrease in the provision of ES, as habitats dry up, water quality declines and flood regulation capacity is reduced (Koundouri et al., 2017). In other cases, the drying of river networks may actually enhance certain ES, such as nutrient cycling and sediment transport, by creating new habitats and promoting biological activity during wet periods. To better understand this relationship, a number of studies have explored the impacts of drying river networks on ES provision and the associated intermittence patterns (Larned et al., 2010).

In terms of water provisioning, results demonstrate that headwater, middle and lowland river domains showed a good agreement between DRNs, although specific results about the number of NPR and PR reaches were related to intermittency patterns (Messenger et al., 2021), natural or anthropogenic factors (e.g. irrigation lands in the Morava catchment) or errors in the delineation of river network and hydrological modelling accuracy (see Künné et al., (2022). Intermittency had almost negligible effects in several years of the series maybe related to a less marked seasonality in the distribution of annual rainfall, i.e. less summer aridity, snowfall and snowmelt processes (Jutebring Sterte et al., 2021), or the existence of buffer effects made by lakes, wetlands and reservoirs. In some occasions, catchment configuration in terms topography, LULC and lithology may control hydrological responses that drive intermittency patterns (Price, 2011). In other occasions there were differences between water provisioning across river reaches, with values that double the median annual value of the series. This imbalance might be related with the different regulatory capacity of the catchment generally devoid of large coverage autochthonous natural vegetation patches, and differences in water provision by precipitation across bioclimatic regions in Europe, which may reduce the provision of water during the dry season. For example, the Guadiaro catchment showed the highest differences between the wet and the dry season compared with the other DRNs, independently of the type or size of river considered what amplifies the severity of droughts with climate change (Spinoni et al., 2021). As explained previously, an spatially-explicit assessment and evaluation of nature-based solutions across the territory such as increase of natural forest area, recovery of floodplains or restoration and generation of ponds and wetlands, that led to the definition of global schemes of blue and green infrastructures networks (BGINs), would represent an adequate strategy to improve the regulatory capacity of the catchments (Staccione et al., 2021) and should be prioritized to face future water security challenges in these regions (Cassin & Matthews, 2021).

Flood regulation refers to the ability of a river network to manage and control the flow of water during periods of high precipitation or melting snow. Natural river systems have evolved mechanisms to absorb and dissipate excess water, such as floodplains and wetlands. These natural features act as buffers, reducing the risk of floods downstream. Climate change, driven by factors like greenhouse gas emissions, can alter precipitation patterns, potentially increasing flood risks in some regions. In turn, landscape configuration and LULC patterns have varying abilities to absorb and retain water runoff. Natural LC types like forests and wetlands have a higher capacity to capture rainfall, slow down runoff and store excess water. On the other hand, urbanization and land-use changes often result in increased impervious surfaces (e.g., concrete, asphalt), reducing the area available for water absorption. This can lead to faster runoff and increased flood risk. When hydrological mechanisms and modification of natural vegetation patterns are not able to control the accumulation of water in certain locations of the riparian areas, a large probability of flooding occur after an event of intense precipitation (Mcnamara et al., 2011). These processes, related to to surface runoff, infiltration, percolation and evapotranspiration differ across space because of landscape configuration (e.g. lower regulation-

capacity in deforested landscapes) and time, in the base of annual cycles of vegetation and climatic conditions. In DRNs, where the flow of water is intermittent or reduced due to various factors (such as water extraction or climate change impacts), the relationship between flood regulation, climate and land cover becomes even more complex. Changes in climate and land cover can exacerbate the drying of river networks, reducing their ability to regulate floods effectively. With reduced water flow, the capacity of the river channel to convey excess water decreases, increasing the likelihood of localized flooding during intense rainfall events. Protecting and restoring natural land covers, preserving floodplain areas and considering climate change impacts are essential for maintaining the resilience of river networks and minimizing flood risks in the face of drying conditions.

Soil erosion referred to the process of detachment and transport of soil particles by water runoff, is primarily influenced by rainfall intensity, slope steepness, soil properties and vegetation cover. Erosion can occur both naturally and due to human activities, and it plays a significant role in shaping landscapes and altering river networks (Issaka & Ashraf, 2017). Climate change can lead to decreased precipitation and increased evaporation rates, resulting in reduced water availability in rivers. As rivers dry up or experience reduced flow, they become more susceptible to soil erosion. In DRNs, climate change also exacerbates the drying trend, leading to reduced water flow and increased vulnerability to soil erosion, while natural vegetation cover, especially trees and other deep-rooted plants, plays a crucial role in stabilizing soil particles, reducing runoff and preventing erosion. When land cover is degraded or altered, such as through deforestation or agricultural processes, soil erosion rates can increase, impacting river networks. Soil erosion, in turn, further exacerbates the drying of river networks (Sponseller et al., 2013). We therefore, contemplated here two different mechanisms leading to ES regulation in DRNs: erosion regulation at source in slopes, on which hillside forests prevents soil loss and avoids the generation of sediment, and erosion transport and filtering, where riparian forests may reduce the amount of sediment delivered to the river network. Sedimentation caused by erosion can reduce the capacity of river channels, increasing the likelihood of channel blockages and reducing water flow. Overall, the interplay between soil erosion, climate and land cover in DRNs forms a feedback loop. Climate change and land cover modifications can accelerate soil erosion, which, in turn, can contribute to the drying trend in rivers.

The spatial distribution of thermal regulation across DRNs has been determined to have a clear dependence of LULC patterns and a recognized morphodynamic control. Climate plays a crucial role in determining the overall temperature patterns in river networks. It encompasses factors such as solar radiation, air temperature, precipitation and wind patterns (Acuña et al., 2008). In areas with a warmer climate, river water tends to be naturally warmer due to increased solar radiation and higher air temperatures. Climate change can exacerbate these effects, leading to elevated water temperatures in river networks. Thermal balance is closely controlled by tree canopies that provide shade that can significantly influence the temperature of river networks. When trees are present along riverbanks, their foliage blocks direct sunlight from reaching the water surface, thereby reducing the amount of solar radiation that heats up the water. This shading effect can help to lower water temperatures and mitigate the impacts of warming. In turn, general landscape configuration across the entire basin affects the thermal regulation of the water across entire valleys and slopes. Vegetated areas, and mainly forests, act as a buffer against excessive heating. On the other hand, if the land cover is dominated by impervious surfaces, it can lead to higher water temperatures due to increased heat absorption and reduced shade (Johnson & Wilby, 2015). In all cases, when river networks dry up, the water volume decreases, and the remaining water is often exposed to direct sunlight. Without tree canopy shading, the lack of water flow can lead to elevated temperatures in the remaining pools or stagnant water bodies. Therefore, maintaining or restoring tree cover along riverbanks can help mitigate the adverse effects of elevated temperatures in regions where drying river networks occur.

Carbon emissions and sequestration, climate and land cover therefore all play interconnected roles in DRNs. CO<sub>2</sub> emissions varied between river basins, with Krka-Butišnica having the lowest evasion per segment and the Morava-Velička the highest in the base mainly of sediment contribution. This is linked to sources of organic and inorganic carbon originating in terrestrial ecosystems and carried underground to rivers (Hotchkiss et al., 2015), which movement may be reduced or even interrupted during low flows or total flow cessation. Additionally, emissions also varied between flow phases although the magnitude of that changes varied between DRNs (Timoner et al., 2012). This refers to a low resistance of autotrophic organisms during dry phase and the higher resistance of heterotrophs to desiccation. Carbon sequestration efforts, such as reforestation or implementing sustainable land management practices, can help mitigate climate change and its impact on river networks. By reducing carbon emissions and increasing carbon storage, these actions can contribute to maintaining adequate water flow and preserving the health and resilience of river systems, as minimizing the risk of drying.

In general, there is a need for considering the dynamic nature of river systems, and specially DRNs, when defining adaptive management strategies that may account for changing environmental conditions. By taking a holistic and context-specific approach to ES provision, it may be possible to enhance the resilience of DRNs and promote sustainable development outcomes.

## **Watershed management for ecosystem function restoration**

The assessment, development and investment on effective watershed management strategies becomes paramount to assure the resilience of ecosystems and the continued provision of services such as water supply, soil conservation and biodiversity in the coming future. In the base of observed results on ES modelling, there are a number of general recommendations for revising watershed management for ecosystem function restoration under drying conditions.

First, there is a need to incorporate climate change projections and drought risk assessments into management plans. Given the predicted increase in frequency and severity of droughts, watershed management plans should consider long-term climate change projections and assess the potential risks of drought to ecosystem functions and services (Vörösmarty et al., 2010). We also need prioritizing the conservation and restoration efforts of natural vegetation. Maintaining and restoring natural vegetation in watersheds is essential to promote water infiltration, reduce soil erosion and support biodiversity. This can be achieved through a range of measures such as reducing deforestation, promoting reforestation and managing invasive species (Rockström et al., 2009). This needs also to be coupled to the implementation of sustainable land use practices that foster conservation initiatives and enhance the ability of ecosystems to adapt to changing climatic conditions. This may include practices such as conservation agriculture, agroforestry and green infrastructure (Jobbágy & Jackson, 2004). The investments in water storage and management infrastructure may actually ensure water availability during dry periods. However, such infrastructure should be designed and managed to minimize negative impacts on ecosystem functions and services (Pielke & Downton, 2000). Finally, it remains crucial to foster stakeholder participation and cooperation. Effective watershed management requires the participation and cooperation of diverse stakeholders in planning, implementation and monitoring of management plans, including local communities, landowners, government agencies, and NGOs.

NBS and integrated catchment management (ICM) of land uses and other anthropogenic infrastructures and activities (hydrological barriers, sewage effluents, water abstraction, etc.) could help regulate river carbon emissions and restore damaged freshwater ecosystem functions and

services (Datry et al., 2021). NBSs refer to the use of natural processes and ecosystems to address various environmental challenges, while simultaneously providing social, economic and environmental benefits. They are designed to work in harmony with nature and harness its inherent resilience and capacity for regeneration. NBS aim to promote sustainable development by conserving and restoring ecosystems, enhancing biodiversity and delivering a range of services to human societies. In a context of ICM for ecosystem services supply in DRNS, NBS can play a crucial role at the watershed scale by controlling the way that land drains water into a particular river or wetland.

The functionality of NBS can adopt different shapes in the base of the ecological process to be restored or the functionality to be conserved (Pérez Rubi & Hack, 2021). In this regard, reforestation of native trees or promoting afforestation dynamics (Chausson et al., 2020) within watersheds can help enhance water infiltration, reduce soil erosion and stabilize riverbanks. Trees also play a critical role in regulating water flow by capturing rainwater, replenishing groundwater and releasing it gradually into streams and rivers during dry periods. NBS also relate to wetland restoration and creation (Keesstra et al., 2018). Wetlands act as natural sponges, absorbing and storing excess water during periods of high rainfall and gradually releasing it during dry spells. Restoring degraded wetlands or creating new ones can help regulate water flow, improve water quality by filtering pollutants and provide habitats for diverse species. Riparian zones management is also essential for preserving water quality, preventing erosion and supporting biodiversity. Planting native vegetation along riparian areas helps stabilize riverbanks, filter runoff and provide shade, reducing water temperature and promoting aquatic habitats. This domain also accepts as NBS a more comprehensive restoration of floodplains, creating natural retention areas and implementing riverbank protection measures that can reduce e.g. the impact of floods and protect downstream areas (Raška et al., 2022). All these NBS should be integrated into comprehensive water resources management plans based upon ICM, considering the entire watershed as a connected system. This approach involves stakeholder engagement, including government agencies, local communities, NGOs, and private sector entities, in order to develop adaptive management strategies and considering the socioeconomic aspects of watershed management alongside ecological goals. It also necessitates policy support, adequate funding, and long-term planning to achieve sustainable outcomes. NBS, connectivity and multipurpose stakeholder engagement for decision making set jointly the foundation of Blue and Green Infrastructure Networks (BGINs).

BGINs creation involves combining ecological patterns and process to enhance ecosystem service supply and biodiversity conservation (Thomas et al., 2022). Blue infrastructure refers to the network of water-related features, such as rivers, lakes, wetlands and coastal areas. These natural water bodies provide numerous ecosystem services, including water purification, flood regulation, habitat creation and recreational opportunities (Chatzimentor et al., 2020). Green infrastructure comprises natural and semi-natural areas like forests and other natural vegetation patches that provide or preserve a range of ecosystem services, such as air purification, carbon sequestration, temperature regulation, biodiversity support and recreational spaces for communities. NBS, as explained above, support the use or mimic natural processes to address societal challenges in an environmentally sustainable manner. They harness the power of ecosystems and biodiversity to provide solutions to issues like climate change and water management. Altogether, BGINs aims to enhance the supply of ecosystem services by integrating and optimizing the benefits of both water-related features and green spaces with multiple purpose, including water management, biodiversity conservation, climate change adaptation, human well-being and in general sustainable development. In a context of flow

intermittence in DRNs and in the base of the results obtained in this work, BGINs can help manage water resources effectively by enhancing natural filtration and retention, reducing the effect of dryness in a context of water scarcity. They can also help regulate water flow, mitigating extreme droughts and reducing erosion risks. BGINs also provide interconnected habitats, allowing species to move and adapt to changing environmental conditions. The combination of blue and green features supports a variety of flora and fauna, contributing to biodiversity conservation and promoting ecological resilience. These infrastructures play also a crucial role in climate change adaptation. They provide natural buffers against extreme weather events like storms, heatwaves and in the context of DRNs, intermittence episodes with the large cascade of accumulative effects. BGINs can also contribute to economic growth by attracting tourism, creating green jobs and fostering sustainable urban planning while optimizing the supply of ecosystem services.

In conclusion, embracing NBSs in integrated BGINs schemes for watershed management presents a promising approach to restore ecosystem functions in drying river networks. By harnessing the power of natural processes and incorporating sustainable practices, we can enhance the resilience of these ecosystems and mitigate the effects of climate change. Through collaborative efforts and informed decision-making, we have the opportunity to protect our water resources, preserve biodiversity and ensure the well-being of both human and natural communities for generations to come.

## Potential effects of climate change in the provision of ES and the design of BGINs

Building climate change scenarios is critical to understand the impacts and cascade effects on the provisioning of ES in DRN. Moreover, comparing current climate conditions to future scenarios can actually be used to show how specific catchment areas or river reaches within a DRN might be more significantly affected on the ES provision they deliver. Following, the approached laid out in this deliverable allows incorporating these new climate change scenarios into the modelling of the 6 ES considered in the second part of the deliverable and analyzing where investments in NBS across the landscape could be more fruitful. Although this is beyond the scope of this deliverable, we will give some hints below on how climate change scenarios could be incorporated.

For example, the water provisioning ES could actually show how different scenarios will reduce water availability in specific areas or seasons. In relation to this, current advances in DRYvER have provided a set of reach-scale daily hydrological projections for the periods 2041-2070 and 2071-2100 for each DRN (Devers et al., 2023). Projections were based on 5 Global Climate Models (GCMs) and 3 shared socio-economic pathways (SSPs) and they give us an insight of the potential consequences of climate change of future water provisioning situations. In this regard, independently of the GCM-SSP combination used, Devers et al. (2023) pointed out a decrease in the annual discharge in all the DRNs meaning a reduction of the water available to maintain the different human activities that directly depend on this resource. This means a high potential risk for water security in these DRN in the upcoming decades. However, Devers et al.(2023) also showed that the reduction of discharge and water provisioning ES is not equally distributed across regions and periods. Accordingly, discharge decreased for all DRNs in spring, summer and autumn while 2 DRNs (France and Finland) showed a slightly increase in winter. Hence, water security risk will be higher in the Spain, Croatia, Czech Republic and Hungary DRNs and especially during the summer months. In the summer months, water availability can be reduced by over 50% compared to the current situation. Moreover, the highest decrease was observed in the far future (2071-2100).

By the same token, the other ES modelled in this deliverable can be modelled with the new climate change scenario datasets (e.g., precipitation, temperature, solar radiation) and differences with the current scenario could be detected spatially explicit across DRNs and their catchments. This is especially significant for flood regulation on hills, erosion regulation, drought regulation, and carbon emissions as their models could be launched with new datasets incorporating these variables. However, the models for flood regulation on floodplains, and thermal regulation do not allow incorporating datasets on climate change scenarios in their current form. For these ES the results should be interpreted as a maximum landscape potential independent of the climatic scenario. Anyhow, the design of specific NBS or the integration of NBS into larger BGINs (i.e. networks of multiple NBS) do not require to have a climate change scenario ran with the current ES models. In fact, the current ES models already pick up which are the areas for restoration (need to increase ES provisioning) or for conservation (need to conserve the current ES provisioning) for specific regulatory purposes (i.e. water, sediments, carbon, etc.).

Finally, it should be noted that the developed digital resources in this deliverable (i.e. datasets, GIS environments and ES models) also allow evaluating the effect of multiple NBS deployed in the landscape. However, a scenario with the deployment of NBS should be generated in order to model the overall effect on the considered ES. Again this is an exercise which is beyond the scope of this deliverable and some adjustments might be needed to evaluate the effects of some NBS within the current ES models, but some insights could be pointed out (see Table 9).

**Table 9.** Potential effect (H: High, M: Medium, L: Low or “-”: null) of Nature Base Solutions identified under the DRYvER project Working Package 5 and the Ecosystem Service Models developed in this deliverable.

Nature Based Solution		Ecosystem Service Models					
		Water p.	Erosion r.	Flood r.	Drought r.	Thermal r.	C-Emissions
Ecosystem / habitat based	Floodplains	H	M	H	M	H	H
	Hillside Forest Cover	H	H	H	H	H	M
	Riparian vegetation	L	H	M	M	H	H
	Hedges	L	L	L	L	M	L
	Buffer strips	L	L	L	L	M	L
	Wood (Hillsides / River)	-	-	-	-	-	-
	Rivers	L	L	L	L	L	H
	Wetlands	M	-	-	M	-	-
	Lakes	M	-	-	M	-	-
Artificial	Small reservoir-pond	L	-	-	M	-	-
	Infiltration (various)	H	-	-	H	-	-
	Bioengineering Erosion	-	H	-	-	-	-
Management	Crop Rotation	-	-	-	-	-	-
	Cattle management	-	-	-	-	-	-
	Water demands	H	-	-	H	-	-
Restoration	Re-meandering	-	-	-	-	-	-
	Bank renaturalization	-	-	-	-	-	-
	Removal of Barriers	-	-	-	-	-	-

In this regard, any action that is restoring or conserving native forests or vegetation (e.g. shrub type in more Mediterranean landscapes) on floodplains, hillsides or riparian areas have a high probability of producing a remarkable effect on most ecosystem services considered in the current developed models. On the contrary, the smaller spatial extent and difficulty of capturing at the current pixel

resolution the effect of hedges and buffer strips might lower down the effect of this habitat types on the current ES models. Among the aquatic ecosystems, rivers have had the largest consideration within DRYvER, but most of the ES modelled are catchment ES that river benefit from, except for the C-emissions (Table 9). In opposition, lakes and wetlands might only contribute to water provisioning or drought regulation (i.e. increasing the available surface water storage). Finally, among the artificially generated habitats, small ponds and actions directed to increase infiltration might only generate a potential effect on Water provisioning and Drought regulation ES results, while bioengineering erosion control measures could have an effect on the erosion regulation results (Table 9). It is important to remark that Water demand regulation could actually be a very management successful measure contributing substantially to Water provisioning, Drought regulation and indirectly to other ES not modelled within this deliverable.

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