Intermittent Rivers and Ephemeral streams: What water managers need to know

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Preface

Intermittent rivers and ephemeral streams (IRES) drain over half the world's land surface and are common water bodies throughout Europe. Often 'hotspots' of regional biodiversity and pivotal for the functional integrity of river networks, many IRES are exploited to achieve growing human demands for water and other ecosystem services. However, suffering from negative perceptions and historically overlooked by researcher compared to perennial rivers and streams, IRES are degraded at alarming rates, and attempts to exclude them from legislations are growing.

In the last two decades, research into the ecohydrology of these prevalent and unique ecosystems has bloomed and management issues have intensified because all climate change scenarios predict expansion in the global extent of IRES. Also, many perennial rivers are gradually becoming intermittent and IRES will become the dominant type of water bodies in the future. Therefore, supported by COST (European Cooperation in Science and Technology, www.cost.eu), we have gathered a dense network of European academics and managers from different disciplines spanning hydrology, ecology, biogeochemistry, and social sciences. This consortium, SMIRES (Science and Management of Intermittent Rivers and Ephemeral Streams, www.smires.eu) aims to compile the scattered knowledge on IRES across Europe for a better understanding of these ecosystems. Although important research gaps remain, our aim was to translate the current level of knowledge to manage, protect, and restore the diverse types of IRES across Europe. This effort resulted in the present handbook, which is the first, to our knowledge, to provide recommendations and guidelines for most aspects related to IRES management issues. Our effort will continue in the near future, notably within the ECOSTAT working group that will integrate IRES into the current management efforts driven by the Water Framework Directive.

Rivers or streams are defined by flowing waters confined within river channels (except during floods) and moving into one direction: rivers are usually larger and deeper than streams, but this is a loose distinction of common usage. The same applies to describing different types of non-perennial flow regimes: "ephemeral" implies a shorter flow duration and lower predictability than "intermittent", but no fixed boundaries exist. Whereas the scientific literature is peppered with attempts to assign names to classes of streams and rivers whose flows cease for varying periods with varying predictability, a global consensus remains elusive and probably will continue to do so. Therefore, rather than opening this semantic minefield, we refer to "intermittent rivers and ephemeral streams" and adopt the acronym "IRES" in this consortium and handbook as a shorthand term for all flowing waters that cease flow and/or dry completely at some point along their course.

We are indebted to the core team, which supervised this handbook preparation, to the many contributors of the different chapters, ant to all the contributors of the working groups of SMIRES, who have done a magnificent job throughout the 4-year timeframe of the action. Last, if this handbook is the first to focus entirely on the management of IRES, there is still much to learn about these dynamic ecosystems and how best to protect their beauty, ecological integrity, and other social values.

May 2020, Thibault Datry, Chair of SMIRES.

Content

1. General Introduction

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Intermittent rivers and Ephemeral Streams (IRES) are river water bodies characterised by temporary flow. Intermittent streams may dry up for some period of time within the year, while ephemeral streams flow only for a small period, usually after rainfall events. Such systems are widespread throughout the world. Intermittence of IRES is characterised by high variability, both in space and time, and can be caused by different reasons, naturally or artificially i.e. dry conditions, freezing of streams, small catchments, water abstraction etc.

The purpose of this handbook is to help water managers to understand the natural processes prevailing in IRES and the importance of this type of streams for biodiversity, but also for local communities. Since it is widely accepted that this type of system has been up to recently neglected, the transfer of knowledge from scientists to water managers for better understanding IRES and the provision of tools for managing them in the best possible way, is considered crucial for their preservation. Furthermore, water authorities need to revise their River Basin Management Plans (RBMPs) in six-year cycles according to the Water Framework Directive (WFD), in which methods to assess ecological status for temporary rivers and suitable measures to protect and/or enhance them will be required.

1.1. Normative framework and international initiatives/trends

IRES are or can be protected and managed under different European directives or international initiatives, which are considered fundamental for the protection of the environment and the sustainability of water resources at European and worldwide level. The importance of IRES is highlighted under the emerging global shifts caused by climate change.

1.1.1. Water Framework Directive (WFD), Directive 2000/60/EC

The Water Framework Directive (WFD), Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000, establishes a framework for Community action in the field of water policy. This directive introduces a new paradigm in water planning and management, since all water bodies should achieve at least Good Ecological Status or Potential, an expression of the quality of the structure and functioning of aquatic ecosystems associated with surface waters.

The ecological status is characterized by biological quality elements, hydro-morphological and physico-chemical elements supporting the biological elements. For rivers, the quality elements are given in Annex V of the Directive and presented in Table 1.1. Member States (MS) developed their own methods/tools for assessing ecological status for these quality and supporting elements.

Table 1.1 Quality elements to be considered for the characterization of the ecological status of rivers

Biological Quality elements	Hydro-morphological elements supporting the biological elements	Chemical and physico-chemical elements supporting the biological elements
Composition and abundance of **Aquatic Flora (Phytoplankton, Diatoms and Macrophytes)**	Hydrological regime - Quantity and dynamics of water flow - Connection to groundwater bodies	General - Thermal conditions - Oxygenation conditions - Salinity - Acidification status - Nutrient conditions
Composition and abundance of **Benthic Invertebrate Fauna**	River continuity	Specific pollutants -- Pollution by all priority substances identified as being discharged into the body of water - Pollution by other substances identified as being discharged in significant quantities into the body of water
Composition, abundance and age structure of **Fish Fauna**	Morphological conditions -- River depth and width variation - Structure and substrate of the river bed - Structure of the riparian zone	

The classification of the ecological status is based on Ecological Quality Ratios (EQRs) which are derived from biological quality values, considering the difference between the current characteristics of the river and the ones that would be found at undisturbed (reference) conditions. The reference conditions are defined for each river type in Annex V of the WFD.

According to the normative definitions of ecological status classifications of the WFD (Annex V of the WFD), at Good Ecological Status *"The values of the biological quality elements for the surface water body type show low levels of distortion resulting from human activity, but deviate only slightly from those normally associated with the surface water body type under undisturbed conditions."*

In order to ensure that the class boundaries are consistent with the normative definitions and are comparable between Member States, a two-phase Intercalibration Exercise was accomplished (Annex V of the WFD). The Intercalibration Exercise was developed at a level of five Geographical Intercalibration Groups (GIGs) across Europe. The Mediterranean GIG for instance, included Bulgaria, Cyprus, France, Greece, Italy, Portugal, Slovenia and Spain. Within the Intercalibration Exercise, a European river typology was developed and all Mediterranean IRES were merged into a single type defined as "Temporary streams" R-M5 Type (Table 1.2).

Table 1.2 Mediterranean river typology as set during the 1st and 2nd intercalibration exercise

Type	River characterisation	Catchment (km^2)	Geology	Flow regime
R-M1	Small Mediterranean streams	< 100	Mixed (except silicious)	Highly seasonal
R-M2	Medium Mediterranean streams	100 — 1 000	Mixed (except silicious)	Highly seasonal
R-M4	Mediterranean mountain streams		Non-siliceous	Highly seasonal
R-M5	Temporary streams			Temporary

The Intercalibration Exercise results were published in the Commission Decision (EU) 2018/229 of 12 February 2018 establishing, pursuant to Directive 2000/60/EC of the European Parliament and of the Council, the values of the Member State monitoring system classifications as a result of the Intercalibration Exercise and repealing Commission Decision 2013/480/EU.

Even so, the data collected for R-M5 type during the Intercalibration Exercise were mainly gathered from highly seasonal water bodies, not extremely temporary or ephemeral streams, due to the fact that biological data from the latter were not available for all Member States. Therefore, the Common Implementation Strategy for the implementation of the WFD has identified, as one of the tasks of its Work Programme for the period 2019-2021, the development of a common methodology for evaluating the status of temporary rivers across the EU. This task is specially focused on those extremely temporary or ephemeral systems that take into account the WFD requirements. This is because, as illustrated by the Mediterranean rivers typology, the WFD missed to focus on the high variability included in "Temporary streams".

The particular characteristics of temporary rivers, their high occurrence in the EU, and the future climate change scenarios urge the adaptation of current biomonitoring methods and the development of new tools to promote an effective and reliable assessment of the ecological status (WFD-CIS, 2019).

1.1.2. Floods Directive (FD), Directive 2007/60/EC

The Floods Directive, Directive 2007/60/EC of the European Parliament and of the Council of 23 October, established that MS should identify the areas at risk of flooding and the corresponding flood risk maps. For such areas MS should develop flood risk management plans focused on prevention, protection and preparedness.

These areas are also associated with IRES, being a very important issue in Mediterranean regions. The high flow regime variability of IRES, with long periods without flow, are responsible for a lack of perception of the risk of flood by the populations, and very often the river bed and floodplain areas are occupied by man. The occurrence of unexpected floods in these areas can pose a danger to the population and cause severe damages on infrastructures, cultural heritage and economic activities. That's why IRES river beds and their flooding areas need to be properly delimited and protected, especially in these watercourses where water rarely flows, and social awareness about likely sudden floods is scarce or absent. Although water flows are scarce or absent for long periods in IRES, their channel and floodplain must be wide enough to accommodate the sudden and high floods that occur.

1.1.3. Habitat and Birds directives, Directive 92/43/EEC and Directive 2009/147/EC

Under the Habitats Directive (HD), Council Directive 92/43/EEC of 21 May 1992 (Art. 3 and 4), Member States designate Special Areas of Conservation (SACs) to ensure the favourable conservation status of each habitat type and species throughout their range in the EU. On the other hand, under the Birds Directive (BD), Directive 2009/147/EC of the European Parliament and of the Council of 30 November 2009 (Art. 4), the network must include Special Protection Areas (SPAs) designated for 194 particularly threatened species and all migratory bird species. Thus, under the latter, IRES habitats would only be included if they happen to host any of the threatened 194 species of birds.

Under the Habitats' Directive the selection of sites, based on scientific criteria to ensure that the natural habitat types listed in the directive's Annex I, are maintained or, where appropriate, restored to a favourable conservation status in their natural range. Thus, in essence, the Habitats Directive focuses on protecting distinct habitats within the river or stream area, rather than on the entirety of the intermittent river or ephemeral stream in itself. Moreover, only two of the just over 230 habitat types that are listed in Annex I include intermittent river formations. These include Habitat 3290 (Intermittently flowing Mediterranean rivers of the Paspalo-Agrostidion) and Habitat 92B0 (Riparian formations on intermittent Mediterranean water courses with Rhododendron ponticum, Salix and others). Nevertheless, there are many other habitat types listed in Annex I of the Habitats Directive that are typically found in IRES. An exercise carried out within the SMIRES COST Network identified fourteen other such habitats that are listed in Table 1.3 below. This table also includes the area covered by these habitat types within the Natura 2000 Network in Europe, together with a brief description of the habitat. These habitats are not restricted solely to IRES but may be found in other freshwater ecosystems such as perennial rivers, estuaries, saltmarshes and around lakes or ponds. Thus, sites that are marked as being mapped as belonging to this habitat, may not be found along an Intermittent River or Stream.

Table 1.3 Habitats of Habitats Directive that are associated with intermittent rivers or ephemeral streams

Annex I Habitat code	Habitat name	Area covered in EU (ha)	Description
1410	Mediterranean salt meadows (Juncetalia maritimi)	83,090	Saltmarshes in the Mediterranean basin dominated by Juncus (rushes) especially Juncus maritimus (sea rush) tolerant of saline soils.
3140	Hard oligo-mesotrophic waters with benthic vegetation of Chara spp.	248,089	Nutrient poor but base rich lakes with Stoneworts (aquatic green algae, *Chara* spp) which often become encrusted with lime.
3220	Alpine rivers and the herbaceous vegetation along their banks	109,090	Rivers in the Alps and other high mountains where the banks are dominated by herbaceous plants rather than trees or scrubs.
3230	Alpine rivers and their ligneous vegetation with Myricaria germanica	3,669	German tamarisk (*Myricaria germanica*) occurs along rivers in the Alps and other mountains growing on silt rich gravel deposits, which are often being destroyed and recreated in floods.
3240	Alpine rivers and their ligneous vegetation with Salix elaeagnos	83,420	Alpine rivers with banks dominated by woody vegetation including rosemary willow (*Salix elaeagnos*), other species of willow (*Salix* spp), birch (*Betula* spp), alder (*Alnus* spp) and sea buckthorn (*Hippophae rhamnoides*).
3260	Water courses of plain to montane levels with the Ranunculion fluitantis and Callitricho-Batrachion vegetation	127,815	Rivers of temperate and northern Europe with floating vegetation often dominated by water crowfoot (*Ranuculus* spp) and other aquatic plants including mosses.
3270	Rivers with muddy banks with Chenopodion rubri p.p. and Bidention p.p. vegetation	38,135	Muddy riverbanks of plain to submontane levels, with annual pioneer nitrophilous vegetation of the *Chenopodion rubri* p.p. and the *Bidention* p.p. alliances.
3290	Intermittently flowing Mediterranean rivers of the Paspalo-Agrostidion	6,317	Intermittently flowing Mediterranean rivers with Paspalo-Agrostidion communities, with the particularity of an interrupted flow and a dry bed during a part of the year.
7220	Petrifying springs with tufa formation (Cratoneurion)	28,236	These are springs with water which is very rich in calcium which forms deposits of tufa or travertine on the vegetation which is often dominated by mosses.

92A0	Salix alba and Populus alba galleries	140,015	Riparian forests of the Mediterranean and Black Sea basins dominated by willows *(Salix alba, S. fragilis*) and Mediterranean and Central Eurasian multi-layered riverine forests with poplar *(Populus* spp), elm *(Ulmus* spp), oak (*Quercus* sp), willows, alder (*Alnus* spp), maple (*Acer* spp), tamarisk (*Tamarix* spp), and common walnut (*Juglans regia*).
92B0	Riparian formations on intermittent Mediterranean water courses with Rhododendron ponticum, Salix and others	6,062	Distinctive, relict thermo- and meso-Mediterranean alder galleries of deep, steep-sided valleys, with *Rhododendron ponticum* ssp. *baeticum*, *Frangula alnus* ssp. *baetica, Arisarum proboscideum* and a rich fern community including *Pteris incompleta*, *Diplazium caudatum*, *Culcita macrocarpa.*
92C0	*Platanus orientalis* and *Liquidambar orientalis* woods (Platanion orientalis)	16,678	Forests and woods, for the most part riparian, dominated by *Platanus orientalis* (oriental plane) or *Liquidambar orientalis* (sweet gum), belonging to the Platanion orientalis alliance.
92D0	Southern riparian galleries and thickets (Nerio-Tamaricetea and Securinegion tinctoriae)	82,313	Tamarisk, oleander, and chaste tree galleries and thickets and similar low ligneous formations of permanent or temporary streams and wetlands of the thermo-Mediterranean zone and south-western Iberia.
9370	Palm groves of Phoenix	1,738	Woods formed by the two European endemic palm trees, *Phoenix theophrasti* and *Phoenix canariensis*. The palm groves of Crete are restricted to damp sandy coastal valleys.

The inclusion of IRES in ANNEX I could help adequately protecting IRES ecosystems across Europe. The focus of the Habitats Directive is on habitats themselves, not on the watershed. This implies that IRES ecosystems might need a better legislative tool to ensure their protection. Moreover, local policy makers tend to prioritize perennial as opposed to intermittent rivers.

1.1.4. European Climate Change Strategy

Climate change is expected to significantly modify the hydrological cycle in rivers and streams in the near future, through global increases in temperature and evapotranspiration, changes in rainfall patterns, and more extended droughts (Hisdal et al., 2001; Schneider et al., 2013). Among them, Mediterranean and semi-arid basins are considered one of the most vulnerable regions with high probability to face acute water scarcity problems in coming years. Observations over recent decades, as well as current global-scale climate change models, indicate changing precipitation and temperature patterns, with an overall increase in the temporal variability and a higher frequency of extreme events such as floods and supra-seasonal droughts (Döll & Zhang, 2010). Hence, the amount and variability of runoff are expected to change significantly according to several future climate scenarios. However, it is difficult to predict, at local and even at regional scale, whether changes in precipitation or evapotranspiration will be greater or lower, whether surface water levels and runoff will increase or decrease, and whether flow variability will change more or less. There are many factors that can affect water levels and flow variability and the effects of climate change on water availability, but it's quite sure that human pressures (water abstraction, etc.) will contribute increasingly to river ecological status deterioration due to its increased vulnerability.

A global-scale analysis on “how is the impact of climate change on river flow regimes related to the impact on mean annual runoff” was analysed in Döll & Schmied (2012). They found that water resources management and climate change are the main drivers altering the spatial and temporal components of flow intermittency. Low regime shifts among perennial, transitional and intermittent regimes indicate strong changes in habitat conditions for freshwater biota and therefore a strong impact of climate change on freshwater ecosystems. Flow regime shifts by the 2050’s may occur on a 6–7% of the world land area, mainly in semi-arid areas as well as in some cold areas. So, in regions that formerly accumulated a winter snowpack, warming temperatures will result in earlier runoff, or a shift from temporary to perennial for those rivers which no river flow discharged in winter months due to freezing. So, mean annual runoff is projected to increase by more than 10% on 50% of the global land area, mainly in northern regions and high mountains (Döll & Schmied, 2012) but, on the other hand, shifts from perennial to intermittent flow regimes will increasingly occur mainly in semi-arid areas and small basins (Döll & Schmied, 2012; Pumo et al., 2016).

Besides changes in flow regime from perennial to intermittent and vice versa, climate change will provide additional effects that will likely damage ecosystem services, which must be taken into account as well for water management purposes. An additional ecosystem response to climate change is an increase in biological productivity, likely affecting the processing of detritus and functioning of the microbial-shredder food web linkage in complex ways. So, changes in production/respiration cycles in rivers will be more significant, which lead to ecosystem alterations, especially in small and temporary rivers. Also riparian vegetation will almost certainly change under future climates (Gosling et al., 2011). More specific and local analyses of climate change impacts on runoff components affecting temporary rivers will be necessary to properly analyse them, like groundwater recharge effect on water permanence (Döll, 2009), or runoff accumulated along the drainage direction. While changes in mean annual runoff are of major interest as they represent changes of total renewable water resources, the assessment of these changes alone is not sufficient for supporting sustainable water management in temporary rivers. So, water managers will be aware of climate change and its effects on temporary rivers, or current perennial rivers that can change in near future.

1.1.5. Sustainable Development Goals

The United Nations has set 17 interconnected Goals under the international initiative of Sustainable Development Goals. These Goals are a call for action by all countries – poor, rich and middle-income – to promote prosperity while protecting the planet. They recognize that ending poverty must go hand-in-hand with strategies that build economic growth and address a range of social needs including education, health, social protection, and job opportunities, while tackling climate change and environmental protection.

Goal 6 is dedicated to Water. Water-related challenges, including limited access to safe water and sanitation, increasing pressure on water resources and ecosystems, disasters and an exacerbated risk of droughts and floods, have received increasing attention in the global development arena. Among the targets of Goal 6, Target 6.6 was set to *"protect and restore water-related ecosystems, including mountains, forests, wetlands, rivers, aquifers and lakes by 2020".* It seeks to halt the degradation and destruction of these ecosystems, and to assist the recovery of those already degraded. The indicator used for whether Target 6.6 is achieved, is the change in the extent of water-related ecosystems over time i.e. spatial extent of water-related ecosystems and inland open waters, quantity of water in ecosystems and quality of water in ecosystems. It is also recommended that countries incorporate a component of ecosystem health in their ecosystem monitoring programme, which is commonly measured through biological indicators. The point of reference for "change over time" is the natural condition, i.e. before large-scale impacts were experienced by the ecosystem.

Under this view, riverine and other aquatic ecosystems are considered very important for achieving sustainability. Ecological monitoring of IRES by developing and using appropriate tools, as well as putting in place proper management strategies for sustaining ecological and hydrological conditions in IRES, becomes even more crucial at local and international level for the provision of water resources and services worldwide.

1.2. Main human pressures on IRES

In the EU, the most common pressures on surface water bodies are hydromorphological pressures (40%) followed by diffuse source pollution (38%), atmospheric deposition (38%), point source pollution (18%) and abstraction (7%) (EC, 2019). Although varying in each country, these are also the main pressures typically affecting IRES, as it is presented in Table 1.4.

Hydrological pressures include construction of dams and weirs, direct water abstraction, but also water discharges from the WWTP and return flows from irrigated areas. Water abstraction is an important key pressure on many water bodies, with a higher regional importance in southern Europe (EC, 2019).

IRES morphology can be modified as a result of straightening and channelization, disconnection of floodplains, land reclamation, dams, weirs, bank reinforcements mainly to facilitate agriculture and protect against flooding.

Point and diffuse source pollution affects water quality of IRES. Agricultural activities result in diffuse emission of nutrients (nitrogen and phosphorus), as well as chemicals such as pesticides. The most important point sources are the industrial and urban effluents insufficiently treated, although over the past few decades, clear progress has been made in reducing emissions from point sources (EEA, 2018).

Table 1.4 presents the processes associated with each main pressure and the chapter of the Handbook where this is discussed. This list is not exhaustive.

Table 1.4 Main pressures affecting processes in IRES

Pressure	*Reasons for the differences between perennial and IRES*	*Impacts*	*Description of processes*
Surface and groundwater abstraction	Exacerbation of the dry phase, the time of permanence and dimension of the pools are affected.	• Changes on species composition and habitats. • Development of alien species	Chapter 2 Chapter 3 Chapter 4 Chapter 5
Water discharges by WWTP, return flows from irrigated areas	Reduction of the timing and duration of the dry phase	• Changes on species composition and habitats. • Development of alien species	Chapter 2 Chapter 3 Chapter 4
Dams and weirs	Social perception of the dry phase. Reduction of water permanence. Changes in water physicochemical characteristics Retention of sediments	• Destruction of aquatic and riparian habitats. • Changes in species composition. • Changes in sediments dynamics. • Erosion phenomena.	Chapter 2 Chapter 3 Chapter 4 Chapter 5 Chapter 6
Vehicular access to the stream bed	Use of river bed during the dry phase as an off-roading trajectory.	• Destruction of the riparian vegetation and morphological changes on river bed and banks, water pollution of pools	Chapter 2 Chapter 3 Chapter 4 Chapter 5
Morphological changes (including the cut of the riparian vegetation)	Removal of riparian vegetation Modification of the river channel Use of river bed during the dry phase.	• Degradation of the riparian zone and habitats • Changes in species composition • Loss of ecosystem functioning. • Floods.	Chapter 2 Chapter 4 Chapter 5
Livestock grazing	Subsurface water keep some vegetation in the stream bed	• Destruction of aquatic and riparian habitats. • Changes in species composition • Increase of nutrient inputs.	Chapter 2 Chapter 4
Diffuse source pollution **Point source pollution**	Less capacity of self-purification	• Peak of pollution during the rewetting phase. • Changes ion species composition.	Chapter 3

1.3. How managers will benefit from this handbook

In view of all of the above, one can conclude that IRES, although up to recently have been highly neglected, their importance is nowadays increasingly acknowledged, especially in the light of climate shift and other emerging worldwide challenges. Therefore, the management of these systems has become a necessity for both the environment and human wellbeing.

Water managers have a great wealth of experience in managing water courses but as already mentioned, the priority given to perennial rivers in combination with poor knowledge of the physical characteristics and processes of IRES, has led to the total neglect of temporary streams.

This handbook is divided into thematic chapters and provides (i) descriptions of the most important natural processes characterising IRES i.e. hydrology and hydromorphology, water chemistry, ecology, (ii) important parameters and challenges related to IRES i.e. ecosystem services and ecological flows, and (iii) some case studies implemented in various countries of Europe for the management of temporary streams (see Table 1.5 and chapter on Case Studies).

By going through this handbook, water managers will gain valuable information and cutting edge knowledge concerning the functioning of IRES. They can benefit from several tools provided for proper and sustainable management of the streams in their area of jurisdiction and will enable them to better contribute to an international effort of preserving IRES, as a crucial component of biodiversity and an important renewable water resource.

Table 1.5 Case studies presenting IRES management and restoration practices. Each case study is presented in detail, in the last part of the handbook.

a/a	Country	River name	Project name	Keywords
1	Cyprus	Yermasoyia river	Hydromorphological restoration and restoration of the riparian zone in Yermasoyia river	Banks restoration Habitats Invasive species
2	Greece	Xrousos river	Xrousos Flood Protection Works	Floods management Protection works
3	France	Clauge river	Restoration of the surface flows of two temporary tributaries of the Upstream Clauge	Restoration Re-meandering Habitats
4	Italy	Torrente Macinino	Rinaturalizzazione con tecniche di ingegneria naturalistica delle sponde del Torrente Macinino	Morphology Restoration River engineering
5	Malta	Wied Il-Qlejgħa	Environmental Restoration of Wied Il- Qlejgħa	Restoration Habitats Riparian zone
6	Portugal	Vascão River	LIFE Saramugo "Conservation of the Saramugo (Anaecypris hispanica) in the Guadiana basin (Portugal)	Species conservation Habitats Restoration

7	Spain	Gaia river	Restoring e-flow in the lower Gaia river	E-flows Flows restoration Dams management
8	UK	River Misbourne	River Misbourne restoration	Chalk streams Morphological restoration Vegetation management

2. Hydrology and morphology of IRES

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2.1 Introduction

2.1.1 In a nutshell

- The temporal patterns of occurrence and connectivity of the habitats define the biologically relevant regime of IRES
- Some specific flow regime variables are highly relevant to better understand IRES and to apply ecological status assessment properly. For example, connected or disconnected pools, time between flow cessation and rewetting, zero flow period are essential
- Regime metrics and classifications must be chosen to be operational and useful to describe habitats and hydrological alteration
- Hydrological information for regime metrics or classifications can be obtained by various sources, not only gauging stations.
- In very dry rivers, hydromorphological indicators can be used to assess quality status.

2.1.2 Why and how to describe hydrology and hydromorphology of IRES?

To preserve or restore aquatic ecosystems, river basin managers seek to describe the characteristics and processes of the IRES hydrology and hydromorphology, supporting physicochemical (chapter 3) and biological processes (chapter 4). This understanding is necessary for several purposes such as (i) to design appropriate sampling protocols and calendars, (ii) to classify the regime of the streams to allow comparisons and the selection of type-specific biological reference conditions and (iii) to identify and evaluate the hydrological and hydromorphological alterations to choose adequate measures where they degrade the ecological quality of the stream.

A short survey was sent in 2017 to water managers from 13 countries across Europe. Nine countries reported existing maps of temporary rivers at different scales, from regional to national. However, there is a lack of quantitative data qualifying the hydrological regimes and their drivers.

Water managers lacked the knowledge of:
(a) how to describe the hydrological regime of IRES,
(b) how flow intermittence extend expansion throughout the hydrographic network,
(c) the scientific soundness of methods used for perennial streams to characterize IRES hydromorphology
(d) methods to monitor hydrology in time and space
(e) how to quantify human pressures on IRES from local water abstractions and climate change,
(f) data about trends on stream temporariness throughout Europe.

In view of the above, this chapter aims to highlight specific features that should be investigated when describing the hydrology and hydromorphology of IRES and to address some of the water manager's needs based on recent scientific methods and knowledge. Some initiatives for monitoring, modelling or classifying different types of IRES are already carried out by some countries and these examples will be described in this chapter. Finally, human activities that are specifically altering stream hydrology and hydromorphology will be discussed.

2.2 Description and characterization of IRES

2.2.1 Hydrological regime – definitions, controls and metrics

There are two approaches, which can be complementary, for tackling the understanding of the hydrological regime of IRES. The first approach considers that IRES are like hydrologically challenged perennial streams and the information on their *flow regime* can be obtained from observed or simulated hydrographs. The second approach considers that IRES are a distinct class of ecosystems (Larned et al., 2010) whose regime includes biologically relevant hydrological conditions not revealed by hydrographs but obtained from other sources of information. These two approaches are used to describe the hydrological regime of 40 types of IRES in 15 European countries in a recent catalogue developed during the COST project SMIRES (Sauquet et al., 2020 - Figure 2.1)

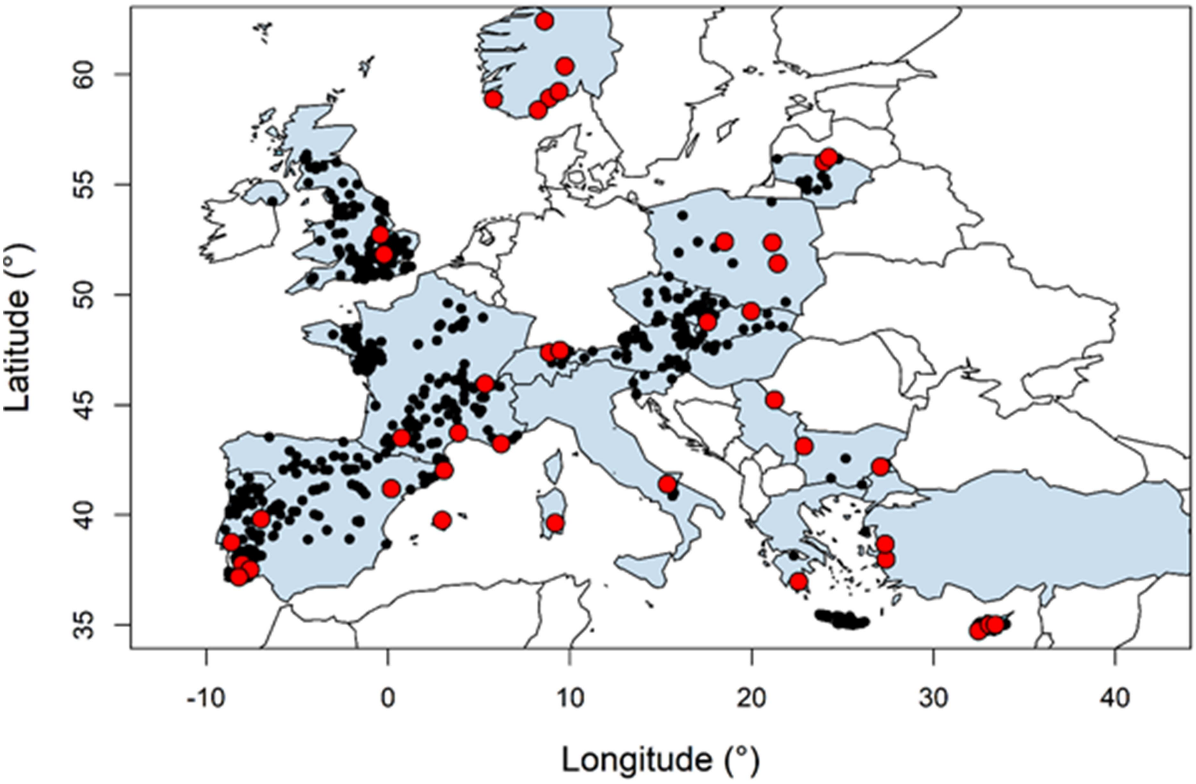

Figure 2.1 Location of the catchments gathered during the SMIRES COST Action. Red dots are catchments presented in the catalogue (Sauquet et al., 2020). Black dots are examples of gauging stations with flow records that met the SMIRES COST Action intermittence criteria. Blue shading indicates countries of members involved in the working group on hydrology of this project.

Following the first approach, the flow regime of IRES can be defined as the temporal variability of its discharge, particularly the quantity, timing and variability in flow. It is generally expressed as the statistical generalizations of hydrological phenomena (e.g. seasonal runoff patterns, median annual discharge, mean and variance of peak flows) at a particular location over multiple years or decades (Thoms and Sheldon, 2000).

The specificity of IRES' flow is that it goes through a zero-flow period, dividing the so-called *wet* and *dry* phases. The timing, frequency, duration, rate of change of these zero-flow periods vary widely between IRES depending on regional and local factors (climate, geology, catchment form, human activities) and influence the riverine ecosystem. The magnitude and rate of rewetting the stream bed also strongly impacts physiochemical and biological processes.

To describe the flow regime, hydrological metrics as indices or statistics computed from multiyear time series of discharge data are used. They are also used for discriminating/classifying different types of regimes (see section 2.4.3). The principal metrics for characterizing the regimes under the hydrograph approach are summarized in Table 2.1.

Table 2.1 Candidate hydrological metrics used to characterize the regimes of IRES calculated from hydrographs (CV - coefficient of variation) reproduced from Costigan et al. (2017)

Hydrological metric	Definition
Frequency of flow-events – zero-flow conditions	
Zero-flow spell count	Mean number of annual, seasonal, or monthly occurrences during which the magnitude of flow remains at or below some threshold defined as zero flow
CV zero-flow spell count	Coefficient of annual, seasonal, or monthly occurrences during which the magnitude of flow remains at or below a threshold defined as zero flow
Duration of flow events – zero flow conditions	
Zero-flow spell duration	Mean duration of annual, seasonal, or monthly occurrences during which the magnitude of flow remains at or below some threshold defined as zero flow
CV zero-flow spell duration	Coefficient of variation in duration of annual, seasonal, or monthly occurrences during which the magnitude of flow remains at or below some threshold defined as zero flow
Number of zero-flow days	Mean annual number of days having a magnitude of flow at or below some threshold defined as zero flow
CV number of zero-flow days	Coefficient of variation in annual number of days having a magnitude of flow at or below some threshold defined as zero flow
Timing and seasonality of flow events – zero flow conditions	
Julian date of annual zero flow	The mean Julian date of the 1-day annual zero flow over all years

CV Julian date of annual zero flow	Coefficient of variation in Julian date of the 1-day annual zero flow over all years
Six-month seasonal predictability of zero-flow periods	Multiannual frequencies of zero-flow months for the contiguous 6 wetter months of the year divided by the multiannual frequencies of zero-flow months for the remaining 6 drier months. Wet and dry 6-month periods are those with fewer and more zero-flow frequencies, respectively
Predictability (P) of zero-flow days	Colwell's (1974) predictability (P) of zero-flow days
Seasonality (M/P) of zero-flow days	Colwell's (1974) seasonality (M/P) of zero-flow days
Rate of change in flow events – before/after zero-flow spell	
Rise rate	Mean rate of increases in flow magnitude (rising limb of hydrograph) over a given time period
CV rise rate	Coefficient of variation in rate of increases in flow magnitude over a given time period
Fall rate	Mean rate of decreases in flow magnitude (falling limb of hydrograph)over a given time period
CV fall rate	Coefficient of variation in rate of decreases in flow magnitude over a given time period
Number of reversals	Number of increases then decreases in flow magnitude over a given time period
CV reversals	Coefficient of variation in number of increases then decreases in flow magnitude over a given time period

The frequency and duration of zero-flow periods are the most common indices used in describing intermittence (Poff, 1996; Knighton and Nanson 2001; Larned et al., 2010; Costigan et al., 2015; Reynolds et al., 2015). These metrics from gauging station records or simulated flows can be automatically calculated using the SMIRES R package (https://github.com/mundl/smires).

The second approach holds that the biologically relevant regime of a temporary stream must be defined as the temporal patterns of occurrence and connectivity of aquatic mesohabitats (Boulton, 2003; Fritz et al., 2006). Following this approach, Gallart et al., (2012) defined the concept of *aquatic states* as the sets of aquatic mesohabitats occurring on a given reach at a particular moment, depending on the hydrological conditions. Six aquatic states were characterized from the wettest to driest: *Hyperrheic, Eurheic, Oligorheic, Arheic, Hyporheic* and *Edaphic* (Figure 2.2). These six aquatic states were too intricate in many cases to obtain adequate information and define operational metrics (Gallart et al, 2017), therefore they can often be simplified into three *Aquatic phases*: flow, pools and dry; corresponding to the three ‘low flow levels’ used in the French Onde observatory (see Section 2.3). There are two major advantages of this second approach for the knowledge and management of IRES: states and phases are defined from a biological perspective and are of a qualitative nature, so that they can be obtained from various sources of information (Section 2.3).

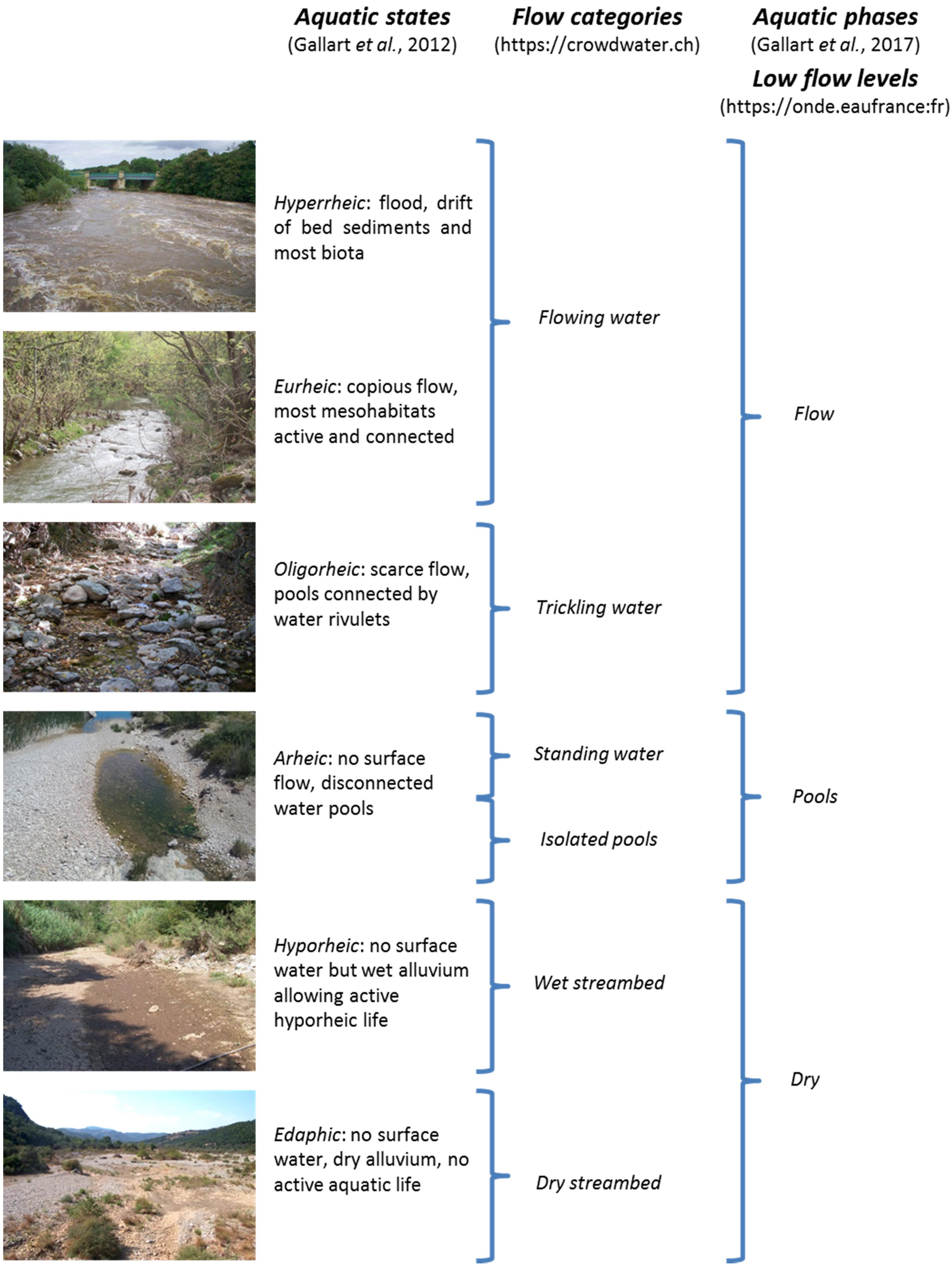

Figure 2.2 Synopsis of the diverse hydrological conditions that can occur in a reach of temporary stream

Using the statistics of these three aquatic phases either at a monthly or seasonal temporal scale, the six metrics shown in Table 2.2 can be obtained. These metrics were selected as they are assumed to be 'ecologically relevant' in the sense of Poff et al. (2010), i.e. they are expected to have some measurable ecological influence.

Table 2.2 Metrics used to characterize the regimes of IRES calculated from information on the three aquatic phases - flow, pools and dry are the source of information. [1] Defined in Gallart et al. (2012). [2] Defined in Gallart et al. (2017).

Acronym	Metric	Definition
Mf	Flow permanence[1]	Long-term mean annual relative number of months with flowing water.
Mp	Pools permanence[2]	Long-term mean annual relative number of months with isolated pools.
Md	Dry channel permanence[2]	Long-term mean annual relative number of months without surface water in the channel.
Sd_6	Predictability of zero-flow periods[1]	Temporal arrangement of no flow periods: the unity minus the relative frequency of the zero-flow months in the wetter 6-month period divided by the relative frequency in the complementary (drier) 6-month period.
ESs	Equinox-solstice seasonality[2]	Temporal arrangement of no flow periods: the relative frequency of 0-flow months in the equinoxes minus that in the solstices.
SWs	Summer-winter seasonality[2]	Temporal arrangement of no flow periods: the relative frequency of 0-flow months in summer minus that in winter.

The first three metrics *Mf, Mp* and *Md* refer to the relative frequencies of the three aquatic phases, so they are complementary and their sum is 1. However, they do not contain information on the temporal patterns.

The third metric, Sd_6, indicates the degree of seasonality, having the value of 0 when zero flows occur equally throughout the year in the long term and 1 when all the zero flows occur in the same 6-month period every year. When the regime is permanent, this metric cannot be computed, so the value of 1 is set to indicate full predictability.

Finally, the two remaining metrics refer to the temporal structure of the zero-flow periods, summer-winter seasonality (SWs), is defined as the difference in the relative frequencies of 0-flow months between summer and winter, it has a value of 1 when there is no flow during summer versus continuous flow in winter and -1 when the opposite occurs. Summer and winter are calculated as for the Northern Hemisphere: this metric would take the contrary sign in the Southern Hemisphere. The metric *ESs,* equinox-solstice seasonality, is defined as the difference in the relative frequencies of 0-flow months between equinoxes and solstices: it takes a value of 1 when there is no flow during equinoxes versus continuous flow in solstices and -1 when the contrary occurs.

2.2.2 Sediment regime

The sediment transport regime of a stream reach (composed of the suspended and the bedload fraction) is imparted by both the hydrological regime and by the sediment supply regime (limited vs unlimited conditions, see Wohl et al., 2015). IRES are characterised by flashy, infrequent runoff events (Gamvroudis et al., 2015) which determine a pulsed sediment transfer dynamics, in which sediment transport takes place only during a very short time every year. However, the amount of the transported sediments (and thus its effects on morphology and biota) depends also on its actual availability. Sediment supply can be considered to be quasi-unlimited in the presence of vast areas of un- or poorly vegetated slopes well connected to the channel network (sensu Cavalli et al., 2013, featuring quite erodible substrates). In contrast, densely forested catchments, highly resistant substrates and/or widespread control works altering sediment connectivity (Marchi et al., 2019) determine limited sediment supply conditions. This latter condition makes the temporal variations in sediment fluxes quite complex because threshold-based dynamics are dominant (e.g. sudden trigger of mass wasting processes delivering pulses of sediments into the channels), in contrast to the more continuous pattern of sediment erosion and transport observed in the case of unlimited supply.

Erosion studies in Mediterranean countries underwent an important push during the last decade and a wide variety of empirical, conceptual or physically based models have been used (mostly in Spain and Italy) to understand erosion and sediment transport processes. These studies are based a) on statistical techniques on field measurements (sediment transport curves), b) erosion plots used in combination with field water and sediment measurements for the quantification of sediment loads, c) use of hydrological models at the field or catchment scale (USLE, SHETRAN, RUSLE, EUROSEM, WEPP, SWAT). The high spatial variability in sediment transport in IRES generates difficulties in quantification of the real erosion and sedimentation rates, especially due to the uncertainties of data scarcity arising from the technical difficulties of obtaining adequate and reliable suspended sediment data. Predicting spatial patterns and intensity of soil erosion and sediment transport can thus be problematic in IRES. It is proved that the calibration of distributed/semi-distributed models by the use of coupled “*Aquatic State river maps*” and *sediment accumulation measurements* in specific river reaches can give consistent simulation results. This approach has great potential in IRES due to the high variability on the sediment yield values within the catchment and that the majority of the annual sediment yield is generated during one single storm event. In Table 2.3 a short review is presented of the most widely known distributed/semi-distributed models used to quantify the sediment mass volumes.

Table 2.3 Distributed models, which incorporates equations to simulate erosion (C=continuous, E=Event-based, Pu=Public). Source : Daniel et al. (2011)

Model	Suited Applications	Main Components	Temporal Scale
AGNPS	Agriculture watersheds	Runoff, infiltration and soil erosion/sediment transport	E

AnnAGNPS	Agricultural watersheds; widely used for evaluating a wide variety of conservation practices and other BMPs	Hydrology, sediment, nutrients and pesticide transport, DEM used to generate grid and stream network	C-daily or sub-daily steps
GSSHA/CASC2D	Suited for both agriculture or urban watersheds; diverse modeling capabilities in a variety of climates and watershed with complex spatial datasets	Spatially varying rainfall; rainfall excess and 2D flow routing; soil moisture, channel routing, upland erosion, and sediment transport	E; C
MIKE SHE	Wide range of spatial and temporal scales; modular design facilitates integration of other models; advanced capabilities for water quality, parameter estimation and water budget analysis	Interception, overland/channel flow, unsaturated/saturated zones, snow-melt; aquifer/river exchange, advection/dispersion of solutes, geochemical processes, plant growth, soil erosion and irrigations	E; C; variable steps
SWAT	Agricultural watersheds; excellent for calculating TMDLs and simulating conservation practices and other BMPs; successfully applied in several watersheds	Hydrology, weather, sedimentation, soil temperature and properties, crop growth, nutrients, pesticides, agricultural management and channel and reservoir routing	C; daily steps
WEPP	Agricultural watershed and analysis of soil erosion of small watersheds	Weather generation, frozen soils, snow accumulation and melt, irrigation, infiltration, overland flow hydraulics, water balance, plant growth, erosion, deposition and residue decomposition	C

2.2.3 Hydromorphology characterization

The EU Water Framework Directive (WFD; European Commission, 2000) introduced the term ‘hydromorphology’, requiring the consideration of any modifications to flow regime, sediment transport, river morphology, and lateral channel mobility to determine the ecological status of natural water bodies. Since then, hydromorphology has increasingly grown as a cross-disciplinary topic at the interface among hydrology, geomorphology, and ecology (Rinaldi et al., 2013). Several definitions of hydromorphology have been proposed and several methods have been adopted for implementing it within the WFD requirements.

Hydromorphology of IRES, as of perennial rivers, is controlled by flow and sediment regimes. To fulfil the WFD requirements, increasing effort has been made to develop methods to assess hydromorphological conditions based on a sound geomorphological approach, with a stronger consideration of physical processes at appropriate spatial and temporal scales. Several classification systems used for describing channel morphology at different scales (geomorphic units or reach scale) have been developed in the recent year. It is fundamental to highlight how geomorphic units and thus the whole reach-scale morphologies are defined based on their topographic and sediment size characteristics scaled by the relevant channel-forming hydraulic variables (bankfull width and/or depth, Comiti and Mao, 2012). In other words, stream morphology at all the scales is invariant with time until relevant bed-material transport or vegetation dynamics modify the channel or the floodplains. Small to intermediate (i.e. before the onset of bedload transport) temporal variations in flow depth and velocity come into play only at the hydraulic unit scale or concur to the definition of the hydromorphological units used for mesohabitat mapping (Vezza et al., 2018).

The River Styles Framework (Fryirs and Brierley, 2005), the SYRAH (Système Relationnel d'Audit de l'Hydromorphologie des Cours d'Eau; Chandesris et al., 2008), the IHG (Indice Hydrogeomorfologico; Ollero et al., 2011), and the method proposed by Wyżga et al. (2010, 2012) are examples of earlier morphological assessment procedures based on a geomorphological approach. Later, in 2010, the Italian National Institute for Environmental Protection and Research (ISPRA) promoted a research program with the objective of developing an overall methodology for the hydromorphological analysis of Italian streams. This methodology, named IDRAIM (Rinaldi et al*.,* 2015), pursues an integrated analysis of morphological quality and channel dynamics hazards aimed at a harmonized implementation of both the WFD and the EU Floods Directive (European Commission, 2007). The Morphological Quality Index (MQI, Rinaldi et al*.,* 2013), which has been later extended to the European scale (Gurnell et al., 2016) is one of the evaluation tools proposed within the IDRAIM framework.

It is important to highlight how both the River Style (Fryirs and Brierley, 2005) and IDRAIM (Rinaldi et al., 2015) frameworks can be fully applied to IRES, as the discipline of fluvial morphology is based on the processes which shape channels and floodplains, notwithstanding how persistent is the water flow. In fact, the ability to access, sample and monitor a dry riverbed at least for some time during the year has favored many studies on sediment transport and channel morphology in IRES. This markedly contrasts with what happened in stream ecology, where IRES have been understudied relative to perennial streams due to their swiftly dynamic and highly diverse environments in which classical “aquatic” metrics and indices fail (see chapter 4 on ecology).

In ephemeral rivers, the use of terrestrial biological communities to assess quality status would be more suitable but knowledge is very scarce as discussed in chapter 4. The use of hydromorphological indices to assess the quality status of these rivers can thus be a more relevant choice than the use of the traditional biological indices

Some research on specific issues in IRES is still needed, especially on the following topics: temporal variation of sediment substrate; bedload transport dynamics (supply versus transport limited conditions during storms of varying magnitude); dynamics of riparian vegetation (areas, locations, size); large wood characteristics (size) and its temporal fluctuation; durations of hydraulic units and link with mesohabitat (see chapter 6 on Eflows).

2.2.4 Hydrological connectivity

In IRES, cessation of surface flow disrupts hydrological connectivity in one or more spatial dimensions, with repercussions for most physical, chemical, and biological processes due to fragmentation. These disruptions in hydrological connectivity are fairly well understood for longitudinal connectivity, but rarely considered for vertical connectivity (the interactions between ground and surface water) which has repercussions for biodiversity.

Boulton et al. (2017) explains the three dimensions of hydrological connectivity as follows (Figure 2.3): "*Longitudinally*, cessation of surface flow halts downstream transport of sediments, other materials, and biota (Hooke, 2003; Rolls et al., 2012), and usually heralds the onset of drying of shallow channel sections, especially riffles. *Laterally*, aquatic habitats on the floodplain and along the riparian zone that were hydrologically linked to the main channel during overbank flows or through bank storage become isolated when water levels decline, interrupting the two-way transfer of energy, sediments, and various organisms (Nakano and Murakami, 2001; Paetzold et al., 2006). *Vertically,* most of the exchange of water between the surface channel and the shallow saturated sediments below (i.e., hyporheic zone, White, 1993) ceases when surface flow stops, impairing processes such as oxygenation of the hyporheic zone by downwelling water or the flux of nutrients to the surface water in upwelling areas (Datry and Larned, 2008; Boulton et al., 2010)."

The spatial and temporal variations of hydrological connectivity in all three dimensions result from flow regime and hydromorphological changes and vary in different types of IRES.

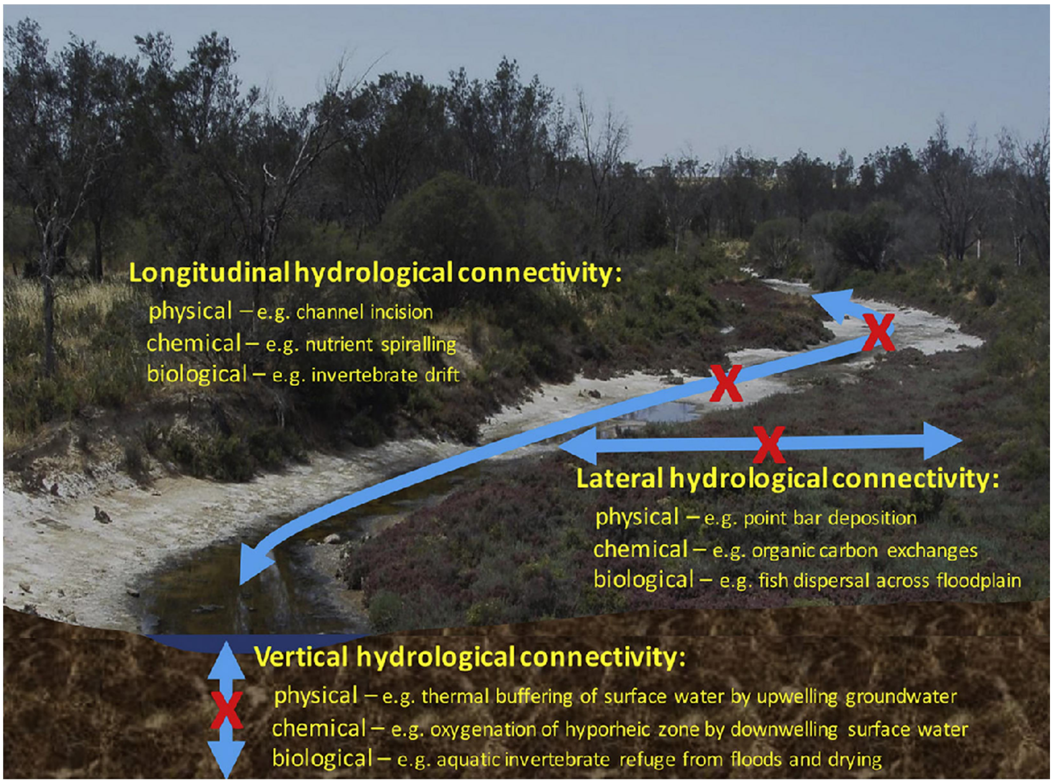

Figure 2.3 Flow cessation and drying in IRES interrupt physical, chemical, and biological processes that rely on hydrological connectivity in three spatial dimensions – longitudinal, lateral and vertical – portrayed as blue lines. The double-headed arrows indicate that many processes operate in both directions, including downstream to upstream (e.g. fish migration). Interruption of hydrological connectivity is indicated by red crosses. (source: Boulton et al. (2017))

Hydrological connectivity can be greatly affected by human activities throughout the catchment (e.g. changes in land use, water abstraction or in-stream works). However, the impacts of the three dimensional hydrological interactions in IRES and their respective influence on ecological and biogeochemical processes are not very well understood and further research is needed.

2.3 Monitoring IRES: how to get information?

Hydrological data to characterize IRES flow regime and connectivity are very scarce because many IRES are ungauged, unmapped, or inaccurately depicted on topographic maps especially intermittent headwaters that are not often officially recognized as proper water bodies.

Current hydrological models can rarely be used as a substitute to obtain information as they usually fail to simulate zero flows. Most of them simulate endless base flows as they do not accurately capture surface-groundwater interactions or local geological peculiarities such as karst. These reasons justify the increasing interest in the development of alternative methods for obtaining the hydrological information where it is needed for investigating and managing IRES.

Hydrological information needed for research and management of IRES can be distinguished in three types:

1. Natural or impaired hydrological regime knowledge is needed to classify IRES and to identify and evaluate the potential hydrologic alteration of the actual stream regimes. This information is usually obtained either using rainfall-runoff models, from old flow records obtained before anthropic regime alterations or from neighbouring unaltered streams in the same physiographic area.

2. Long- or medium-term information on the actual hydrological regime is needed to classify the IRES, to design adequate sampling calendars, to define the reference biological conditions for evaluating the biological quality, and to characterize and assess the potential hydrologic change or alteration. Nevertheless, the question arises as to whether biological reference conditions should be selected taking into account the natural or actual hydrological regime when it is altered.

3. Real-time or recent information on the active aquatic states or phases when biological samples are taken is necessary for adequately interpret the aquatic life found. This kind of information is particularly needed for research (e.g. Pařil et al., 2019) and monitoring purposes but it may be also necessary for some operational targets.

2.3.1 Gauging stations

Gauging stations are the most operational and standard method for monitoring stream hydrology but in most countries they are rarely located in IRES. Their location is usually chosen to integrate a certain amount of flows or close to urban areas or places of economic interest and thus headwaters are poorly gauged.

This leads to underestimate the regional extent and distribution of flow intermittence (Snelder et al., 2013). Discharge data from one gauging station will give information for a specific location but not on the spatial extent of intermittence.

Another serious issue related to the utility of gauging stations for monitoring IRES is that they are designed to measure flowing water but not to capture zero-flows events or the stream bed conditions after flow cessation. Nevertheless, when gauging stations are located in alluvial channels and relevant water seepage under the gauging control is identified, cut-off structures are frequently used to interrupt the hyporheic flow and measure it as surface flow (WMO, 2010). This means that in many cases the real cessation of surface flow occurs for some discharge threshold above 0 in the record, while lower discharges actually correspond to the pools phase. This is an inconvenience for generalizing zero-flows in large sets of gauged records (Tramblay et al., submitted), but it may be an advantage at the station level because it provides information on the occurrence of the pools phase (Gallart et al., 2012; 2017).

Data from gauging stations remain valuable as they are often continuous, exist for long periods and are able to provide information both at the long-term and real-time scales. Nevertheless, they have the issues of location and information quality stated above, along with other error causes for the zero readings (Zimmer et al., 2020). Therefore their information must be combined with other kinds of data for assessing IRES and to improve temporal and spatial extrapolation when modelling intermittence (see section 2.4.1).

A database of 452 discharge time series of IRES was built within the framework of the COST project (Tramblay et al., submitted). This database contains rivers characterized by natural or moderately influenced regime, and catchment areas smaller than 2000 km² (figure 2.1). The absence of dams or reservoirs upstream of the gauging station was verified. The interactive map with the metadata for every gauging station is available at https://drive.google.com/open?id=16oqeQgGhW1J6R8uOWBV7vM5Bw5A&usp=sharing.

Some alternative approaches to monitor and to characterize the regime of IRES are thus developed, with increasing technical quality and popularity. They are described in the following paragraphs.

2.3.2 Real-time or deferred visual observations

Direct observations made by individuals are the best way to obtain information on the 6 aquatic states or 3 phases described in Figure 2.2 When these observations are documented at the time when they are made, they form real-time records and can be also used as long- or medium-term records if systematically repeated during a sufficiently long period. Alternatively, interviews of local stakeholders (citizens or water professionals) can be conducted to understand the temporal patterns of the aquatic phases in a stream during the preceding several-year period. Unfortunately, these interviews cannot provide the flow state and time resolution of the documented direct observations but can be useful in estimating the regime metrics shown in Table 2.2 (Gallart et al., 2017).

In France, a national observatory to monitor low-flow levels called “Onde” (https://onde.eaufrance.fr/) was set up in 2012 and is maintained by the French Biodiversity Agency (OFB). This observatory was designed with two key objectives: to be a stable network of knowledge on summer low-flow levels and to help in anticipating and managing water scarcity. It consists of 3300 stations spread all over France, representing different climate and geographical context, and in areas that were not extensively monitored, especially headwaters. OFB staff visually assesses the river flow level around the 25th of each summer month, from May to September. Three main descriptors corresponding to the three phases of Figure 2.2 are used:

- Visible flow : water can be seen to be flowing continuously
- No visible flow : water is present, maybe in the form of pools, but no streamflow can be seen
- Dried out : there is no water

One-off campaigns are carried out by OFB staff at higher frequency (weekly) or outside of the summer months when decided jointly by local water stakeholders. The use of a harmonised observation protocol across France since 2012 means there is now a set of comparable data over a seven-year period. In a recent modelling study, these flow surrogates were combined with discharge data and groundwater levels to extrapolate temporally and spatially intermittence (see section 2.4.1).

2.3.3 Citizens apps

Citizen science and crowdsourcing can be used to expand professional networks and for large geographical areas. Crowdsourcing can provide information on the hydrological state of IRES (e.g., dry, standing water, or flowing water) anywhere people regularly pass-by these streams, e.g., while they are hiking, on their way to work or school, walking the dog.

An additional advantage of citizen science projects is that they raise awareness of environmental issues (Johnson et al., 2014). This could be crucial for increasing the awareness and appreciation of citizens for IRES, and thus their willingness to protect them.

CrowdWater is one of the citizen science projects in which data on IRES are collected ((https://crowdwater.ch/en/welcome-to-crowdwater/). The freely available CrowdWater app can be used to record observations on the state of IRES, as well as relative stream water levels, soil moisture and plastic pollution. The GPS in the mobile phone is used to determine the location for new observations (called "spots") and to find existing locations for repeated observations. Anyone can set up a new observation location or contribute observations to existing locations. The citizens do not only choose the category that best describes the state of the IRES but also upload a picture of the stream. These pictures can later be used for data quality control or other analyses (e.g. automated identification of the flow conditions using artificial intelligence). All data submitted via the app are publicly available.

Although the CrowdWater app was first released in April 2017, the IRES category wasn't promoted until April 2019. By 27/01/2020, 4533 observations had been submitted for 1480 locations, including 187 locations with five or more repeated measurements. For one location, there were 215 repeated observations. Figure 2.4 provides an example of time series of observations for an intermittent stream in Portugal. As is typical for citizen science projects, the majority of the observations are submitted by a small group of dedicated volunteers. For the CrowdWater project, the top 10 most active volunteers in the IRES category submitted 63% of all the observations.

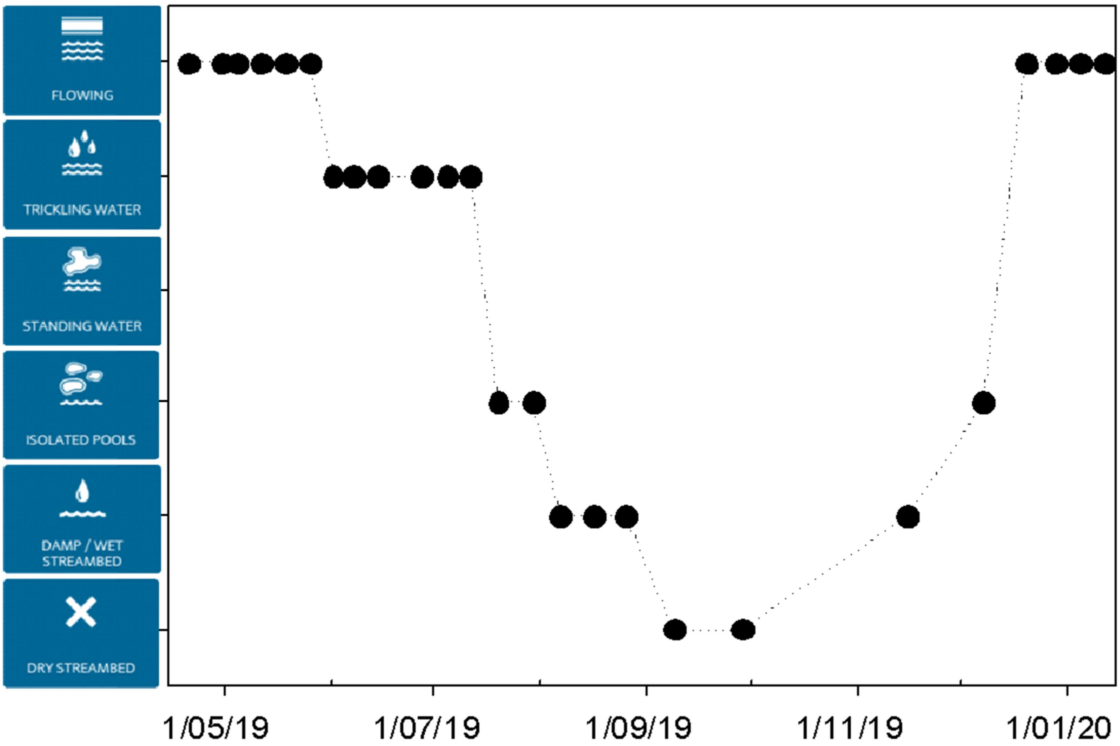

Figure 2.4 Example of time series of observations on the state of an IRES in Portugal collected with the CrowdWater app. Source : https://www.spotteron.com/crowdwater/spots/89088

Enquete d'Eau, Streamtracker and RiuNet are other citizen science projects on IRES and focus on IRES in France, the USA and Spain, respectively (see Table 2.4). Similar to CrowdWater, the citizens participating in these projects can decide on where to make their observations. RiuNet guides the citizen not only through the assessment of the hydrological status of the stream but also the ecological status by gathering more information (e.g riparian area, degree of alteration of the stream…). Loiselle et al. (2016) found that for the project FreshWaterWatch, citizen scientists tended to make more repeated measurements if they got to choose the location for which they wanted to contribute data themselves rather than when a location was assigned to them. Other citizen science projects focus on repeated measurements on specific dates along certain rivers. For example, a citizen science project in the USA has collected data on the presence and absence of surface water along three rivers in the lower Colorado basin (each longer than 30 km) for a specific weekend in June for more than 10 years (Allen et al., 2019).

Table 2.4 Overview of four citizen science projects to monitor the state of IRES

Project	**CrowdWater**	**Enquete d'eau**	**Streamtracker**	**RiuNet**
Geographic area	Worldwide	France	US (worldwide)	Iberian Peninsula

				(Worldwide)
Funding	Swiss National Science Foundation/University of Zurich	L'Agence Française pour la Biodiversité (AFB)	Citizen Science for Earth Systems Program (NASA)	FECYT, EU LIFE programme.
Website	CrowdWater.ch	enquetedeau.eaufrance.fr	streamtracker.org	riunet.net
Start of project	2017	2017	2017	2017 (for hydrology)
Data collection	App (but also possible via the website)	Website	Website	App
Classes	1. flowing water 2. standing water 3. connected pools 4. isolated pools 5. wet streambed 6. dry streambed	1. flooding 2. visible flow 3. weak visible flow 4. non-visible flow 5. dry	1. flowing 2. not flowing	• flowing water • isolated pools • dry riverbed
Photo	Yes	optional	optional	yes
Number of locations on 27.01.20	1480	846	939	405 for hydrology (from a total validated of 1060)
Number of observations on 27.01.20	4533	1134	4918	405

Both professional observation networks and citizen science are promising tools to advance the knowledge and mapping of IRES, compensating for the feature and spatial limitations of the network of gauging stations. Software tools like TREHS (see section 2.4.2) allow the use of these data for calculating metrics and classifying the regime of the monitored streams but an effort should be made for coordinating efforts, updating the quality of the observations made and ensuring their optimal exploitation for research and management purposes.

2.3.4 Field loggers

Field loggers with sensors such as electrical conductivity, water temperature, floating switches or propellers may inform on the presence-absence of water and detect changes in hydrological conditions. Time-series data from the loggers can be used to track the movement of wetting and drying fronts (Bhamjee and Lindsay, 2011) and the persistence of surface waters in different reaches (Vander Vorste et al., 2016). These data can then be translated into hydrological metrics to assess and compare hydrological regimes.

Thanks to their relatively low prices (Assendelft and van Meerveld, 2019; Chapin et al., 2014), arrays of multiple sensors and loggers can be placed within and across stream network to spatially describe flow and water presence-absence regimes and connectivity in space and time (Jaeger and Olden, 2012; Vander Vorste et al., 2016). These arrays are especially useful if the study areas are remote and frequent visitation is impractical. However, there are several limitations to using loggers. For example, in-stream and riparian loggers may be washed out or buried during flood events and subject to vandalism. Therefore, regular maintenance and data downloads are required, particularly where flow regimes are flashy or risks of vandalism are high. Most sensors are unable to differentiate flowing or standing waters.

Sometimes detecting the presence of water is difficult; if moist sediment builds up on the sensor probes, the sensors interpret that as wet conditions. Several diverse sensors are recommended, combined with regular site visits to ground truth the sensor datasets. An alternative to improve the efficacy of the sensors arrays is to combine them with time-lapse cameras (Assendelft and van Meerveld, 2019; Straka et al., 2019) so the quantitative data recorded by the sensors is supplemented with the qualitative information captured by the photographs (Marek Polášek, personal communication, 2020).

2.3.5 Remote-sensing data and aerial photographs

Remote-sensing data may provide efficient deferred or near-real time proxies for the hydrologic conditions of IRES and their floodplains. Callows and Boggs (2013) used the Moderate Resolution Imaging Spectroradiometer (MODIS) sensor with high temporal frequency (twice daily) and 250 m of spatial resolution to derive five flow types and six hydrological metrics in an entire 1996 km² dryland area in northern Australia. The COPERNICUS program managed by the European Space Agency, especially images from Sentinel 1 and 2 satellites, can also be used to visualize changes in the flow regime on near-real time, with a time resolution of 5 days. Unfortunately, this resolution is not suited for headwaters because of images resolution issues and vegetation density. Nevertheless, Marcus and Fonstad (2008) successfully used optical remote imagery at 4m (IKONOS satellite) and 1 m airborne resolutions to map in-stream habitats.

Gallart et al. (2016) used series of orthophotographs with resolution of 0.25 and 0.5 m available on the website of the Cartographic and Geological Institute of Catalonia (ICGC) for visually obtaining deferred operational statistics of the three aquatic phases (Figure 2.2) in a sample of relatively small streams. The same kind of information may be obtained using Google Maps images, assisted by point information obtained from Earth-View recurrent shots taken from bridges (Gallart et al., 2017)

Imagery using light aircraft and drones is more suitable for repeated mapping of intermittence in smaller streams, although its utility is more focused to research. For repeated imagery at different time or date, such as time-lapse photography can be used to characterize real-time temporal variation of the hydrological conditions in sections of small IRES for either research or operational purposes (Puckridge et al., 2000).

2.4 Classifying regimes of IRES: how information can be managed?

Following Uys and O'Keefe (1997), classification of stream regimes is needed to assist in standardizing definitions of the range of regimes encountered and in promoting common use and clear communication of descriptive terms in a multidisciplinary environment. Regime classification allows assigning streams or stream sections to a particular type, so the relationships between ecological metrics, regime and regime alteration may be analysed (Poff et al., 2010). For operational purposes, the regime classification should be defined using threshold values of metrics obtainable from available information.

2.4.1 Prescribed classifications

Some European Member States implemented classifications of the IRES according to their natural regimes in the respective transpositions of the WFD ("ORDEN ARM/2656/2008" in Spain and "DECRETO 16 giugno 2008, n. 131" in Italy); the example of Spain is shown in Table 2.5.

Table 2.5 Definition of the different stream types in terms of temporariness in the Spanish transposition of the WFD (ORDEN ARM/2656/2008). The data used for this classification are the flow series simulated as a natural regime with the help of a rainfall-runoff model

	Definition
Perennial	Water courses have natural flow regime conditions flow during the whole year.
Temporary or seasonal	Water courses where natural flow regime conditions present a marked seasonality, showing reduced flow or dry riverbed in summer, and flow is present during an average period of 300 days in a year.
Intermittent or strongly seasonal	Water courses where natural flow regime conditions present a high temporality, and flow is present during an average period between 100 and 300 days in a year.
Ephemeral	Water courses that in natural flow regime conditions only flow sporadically, mainly in storm episodes, during an average period less than 100 days in a year.

This classification is operational, it is obtained for a 65-year period from the flow series simulated as for a natural regime with the help of a rainfall-runoff model (https://www.miteco.gob.es/es/cartografia-y-sig/ide/descargas/agua/simpa.aspx). It is also environmentally daring, because the streams are classified according to their intended natural regime.

Nevertheless, the main inconvenience of this classification, as well as most other classifications, is that it does not take into account the occurrence of pools (connected or disconnected) after the cessation of flow. However, there is consensus among authors that aquatic life in IRES depends not only on the occurrence of flow but also on the presence of surface water in the form of stagnant pools when flow is interrupted (e.g. Robson et al., 2013; Davis et al., 2003). Some pools may persist through months of no rainfall whilst others may change in size or disappear, for reasons not always easy to be identified (Seaman et al., 2016; Bourke et al, (submitted)). Consequently, there are some terminologies and classifications of the regime of IRES that mention the occurrence of pools, but fail to operationally include their frequency in the identification of regime classes due to the lack of adequate statistics (e.g. Uys and O'Keeffe, 1997; Rossouw et al., 2005; Gallart et al., 2012).

2.4.2 TREHS classification on the basis of the flow-pools-dry metrics

A regime classification for stream reaches was designed by Gallart et al. (2017) that was intended to be i) operationally applicable from simulated or actual available information not only at gauging stations, ii) taking into account the metrics of the three aquatic phases (flow, pools, dry), iii) able to be represented in a single graph, iv) conflict-free from the most usual terminologies, and v) defined from hydrological features assumed to have biological implications, though these are not yet proved. However, practical reasons made it appropriate to discard the representation of the temporal structure of the aquatic phases represented in the Sd_6, SWs and ESs metrics (Table 2.2). Therefore, it is to be expected that the biological significance of the classes designed will have different implications in distinct climate settings.

The approach selected for regime classification was based on the Flow-Pools-Dry plot (Figure 2.5), using the following attributes:

- Perennial: Permanently flowing, except on rare occasions.
- Fluent: Usually flowing.
- Stagnant: Usually takes the form of isolated pools.
- Alternate: Rotates between the three aquatic phases.
- Occasional: Stream usually dry that sometimes, but not often, has flowing or stagnant water.
- Episodic: Dry stream with either flowing or stagnant water at infrequent intervals.

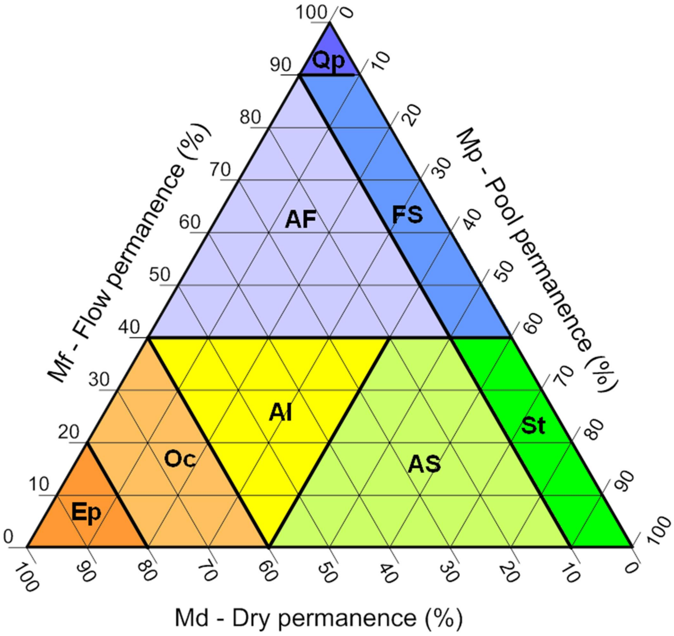

Figure 2.5 Distribution of the TREHS regime classes in the Flow-Pools-Dry plot (Gallart et al., 2017). Qp: Quasi-perennial; AF: Alternate-Fluent; FS: Fluent-Stagnant; St: Stagnant; AS: Alternate-Stagnant; AI: Alternate; Oc: Occasional; Ep: Episodic. The three metrics (triangle altitudes) are from the bottom to the top Mf: flow permanence ; from the left side to the right vertex Mp: pool permanence and from the right side to the left vertex Md: dry channel permanence.

These terms are combined to identify nine types of regime, as shown in Table 2.6 and Figure 2.5, where the threshold values defined for the three metrics are indicated. Two of these boundaries are assumed as the most relevant for aquatic life: Mf smaller than 0.4 is assumed as a practical boundary where usual WFD methods cannot be used to assess biological status; and Md smaller than 0.1 represents conditions with quasi-perennial surface water, either flowing or stagnant. Nevertheless, these values as well as the threshold metrics indicated in Table 2.6 are a provisional suggestion that should be updated for diverse regional settings with biological information.

Note in Figure 2.5 that a classification based solely on the permanence of flow (Mf) would place all the classes just on the left side of the triangle and would therefore similarly classify streams that never dry out (located along the right side of the triangle) with those that recurrently dry out (located along the left side of the triangle).

Table 2.6 Nomenclature and metrics boundaries of aquatic phases regimes as used in the TREHS regime classification. Mf: flow permanence, Mp: pool permanence; Md: dry channel permanence. The characteristic metric boundaries used for defining the regimes in Figure 2.5 are shown in bold.

Regime	*Mf*	*Mp*	*Md*
Perennial (Pe)	0.99<*Mf*≤ 1.00	0.00≤ *Mp*<0.01	0.00≤ *Md*<0.01
Quasi-perennial	**0.90<*Mf*≤**	0.00≤*Mp*≤	0.00≤*Md*≤

(Qp)	0.99	0.10	0.10
Fluent-Stagnant (FS)	$0.40<Mf\leq 0.90$	$0.00\leq Mp<0.60$	$0.00\leq Md<0.10$
Alternate-Fluent (AF)	$0.40<Mf\leq 0.90$	$0.00\leq Mp<0.50$	$0.10\leq Md<0.60$
Stagnant (St)	$0.00<Mf\leq 0.40$	$0.50\leq Mp<1.00$	$0.00\leq Md<0.10$
Alternate-Stagnant (AS)	$0.00<Mf\leq 0.40$	$0.40\leq Mp<0.90$	$0.10\leq Md<0.60$
Alternate (Al)	$0.00<Mf\leq 0.40$	$0.00\leq Mp<0.40$	$0.20\leq Md<0.60$
Occasional (Oc)	$0.00<Mf\leq 0.40$	$0.00\leq Mp<0.40$	$0.60\leq Md<0.80$
Episodic (Ep)	$0.00<Mf\leq 0.20$	$0.00\leq Mp<0.20$	$0.80\leq Md<1.00$

2.4.3 Hydrological and hydrodynamic modelling

Numerous models have been developed by researchers and engineers to study surface water processes in perennial rivers (e.g. Pilgrim et al., 1988; Wheater et al., 2007) and discharge in response to precipitation patterns. Hydrological models have the potential advantage to provide estimations at ungauged locations and future river flows under global change context.

The choice of the type of models depends on the use of the model and on the data availability. Lumped models are well suited for a global assessment of water resources. They are usually preferred due to their ability in describing river flow patterns without an explicit representation of the catchment processes but are unsuited to identify key drivers of flow intermittence among catchment characteristics.

The semi-distributed hydrological SWAT model (Soil Water Assessment Tool; Gassman et al., 2007) has been used - with modifications or couplings - to simulate daily discharge under natural, current, and future conditions (Jaeger et al., 2014; Tzoraki et al., 2016) and aquatic states of IRES (De Girolamo et al., 2016). The regional hydrological model

SIMGRO model was applied in Evrotas river (Greece) to provide flow data subsequently interpreted in terms of ecological status (Querner et al., 2016).

Dean et al. (2016) and Azarnivand et al. (2020) have applied the integrated surface–subsurface physically based hydrological model CATchment HYdrology (CATHY) to small catchments in Australia. CATHY was calibrated against observed river flows and groundwater levels. Dean et al. (2016) have pointed out that human-induced land use changes have affected groundwater storage more than runoff pattern, which is consistent with field observations. Azarnivand et al. (2020) have analysed the sensitivity of discharge (annual flow, flow duration, runoff coefficient, fraction of days with flow) to precipitation patterns (rainfall frequency). Niedda and Pirastru (2014) have the physically based distributed hydrological model SSFR (Saturated Subsurface-Flow Routing model) to study the non-linearity of the rainfall–runoff processes.

At the local scale, hydrodynamic models for river network can provide a detailed description of hydraulic conditions in the channel bed (e.g. water depth, depth-averaged flow velocities) and hydrological processes (e.g. flush events, pools formation, transmission losses) occurring in temporary streams (Figure 2.6). They are found useful to inform on interaction between surface and subsurface flow as well as on influencing ecological processes (Trancoso et al., 2009; Tzoraki et al., 2009; Theodoropoulos et al., 2019). Conversely, using a global hydrological model (e.g. Döll and Schmied, 2012) is certainly hazardous to study flow intermittence at the fine resolution (i.e. in headwater streams).

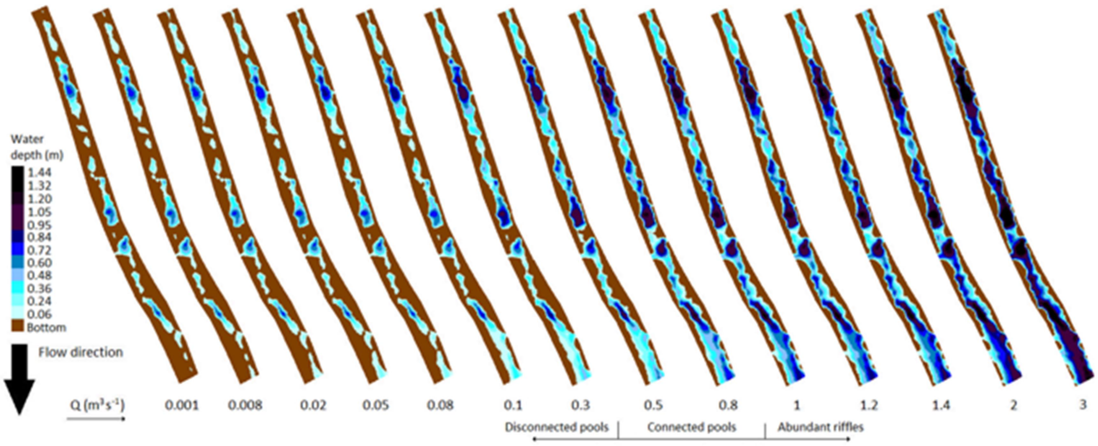

Figure 2.6 Water depth simulation to facilitate the discrimination between the different aquatic states (from Theodoropoulos et al. (2019)).

However applying rainfall-runoff models to intermittence streams is not straightforward for several reasons:

- Accurately simulating extremes, including floods and severe low-flow events that may lead to zero flows, is always a challenging task;
- Models should ideally represent the effects of surface–groundwater interactions or local geological peculiarities on river flow regime, complex nonlinear processes in runoff generation, such as deep groundwater and antecedent moisture conditions antecedent to rainfall events (Ye et al., 1997). A suitable formulation of the recession phase to allow streams to completely dry out is also required to expect a good match between simulation and observation (Ivkovic et al., 2014);

- For ephemeral streams, the number of zero-flow events can be quite large even for a long-term monitored site and gauging stations give no information on moisture conditions during periods of no flow. Thus, parameter estimates related to rainfall-to-runoff transformation can be highly uncertain (Croke and Jakeman, 2007); Most of them do not consider the formation of isolated pools and dry riverbeds that play a significant role in the hydrological and biogeochemical processes.

Improving hydrological models to gain in performance and developing new ones is still an ongoing task.

Successive versions of the lumped conceptual IHACRES model have been tested in arid and semi-arid regions (Ye et al., 1997, 1998; Croke and Jakeman, 2007; Ivkovic et al., 2014). These applications on numerous catchments in Australia have supported the need for a high level of complexity in conceptual models, and for improvements to reach the performance usually obtained in temperate, humid catchments. Ivkovic et al. (2014) have included an explicit representation of groundwater storage and a linear relation between storage and baseflow that replaces the classical slow transfer function. Viola et al. (2014) have developed a new conceptual lumped model dedicated to the simulation of daily streamflow in semi-arid areas. The model parameters are related to soil and vegetation characteristics to facilitate the application to ungauged basins.

To address the current limitations of the models in simulating no-flow conditions, a possible option is to apply a post-processing technique. The post-processing technique is a convenient way to simulate zero-flow events without modifying the structures of the models.

Cipriani et al. (2014) have used a quantile-mapping approach (e.g. Snover et al., 2003) and spot gauging data, to correct the outputs of two rainfall-runoff models to finally simulate zero flow events of an intermittent river in south-eastern France. Facing the inability of models to simulate zero flow, the simplest approach consists in defining cut-off values equivalent to zero (e.g. on the outputs of the SWAT model in Levick et al. (2018), and of the water balance model in Yu et al. (2018).

Lastly, developing models taking advantage of new data is another option. Citizen science creates opportunities to overcome the lack of hydrological data and to develop new models combining data from conventional observation networks and citizen science initiatives. An example is given by the approach suggested by Beaufort et al. (2018, 2019) relating discrete field observations of flow intermittence to continuous daily discharge and groundwater-level data. Empirical models have been developed first to predict the daily probability of intermittence at the regional scale across France and second to reconstruct local drying dynamics at each ONDE site (Figure 2.7).

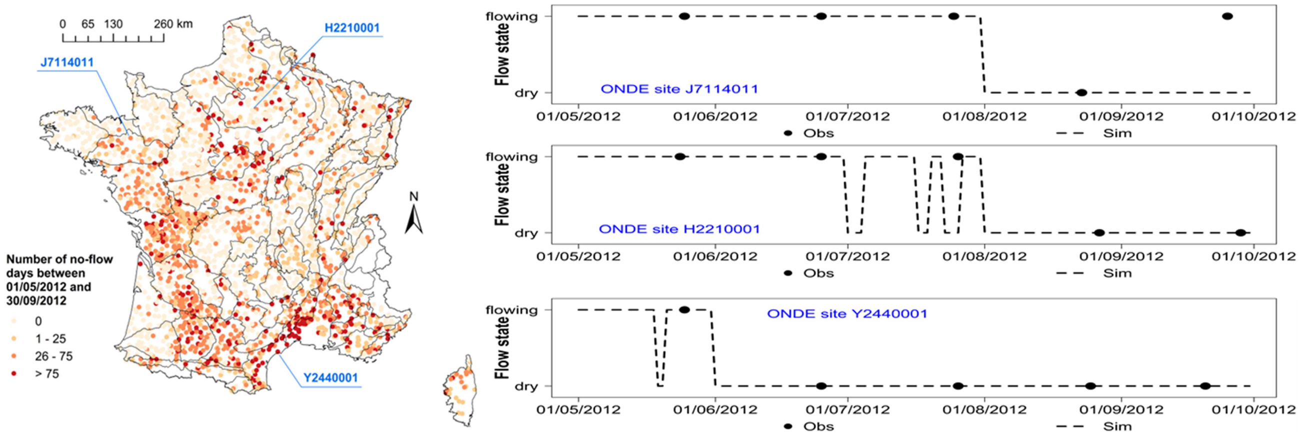

Figure 2.7 Statistics and time series of flow states for the year 2012 simulated by artificial neural network and using discrete observation from the ONDE network, according to the approach developed by Beaufort et al., (2019). Results are obtained for 3080 ONDE sites (https://onde.eaufrance.fr/) displayed on the map.

2.4.4 Trends

Climate change is likely to increase the proportion and the repartition of IRES across Europe. Water managers are thus interested in knowing the already observed trends of intermittence in the previous decades and the projected trends. A recent study carried out by Tramblay et al. (submitted) analysed the trends of annual and seasonal number of zero-flow days, the maximum duration of dry spells and the mean date of the zero-flow events, using a database of 452 rivers in European and in Mediterranean countries outside Europe, with varying degrees of intermittence. In addition, the relationships between flow intermittence and climate are investigated using the Standardized Precipitation Evapotranspiration Index (SPEI) and six climate indices describing large scale atmospheric circulation. The results indicated a strong spatial variability of the seasonal patterns of intermittence and the annual and seasonal number of zero-flow days, which highlights the controls exerted by local catchment properties. Most of the detected trends indicate an increasing number of zero-flow days which also tend to occur earlier in the year, in particular in Southern Europe (Figure 2.8; Figure 2.9).

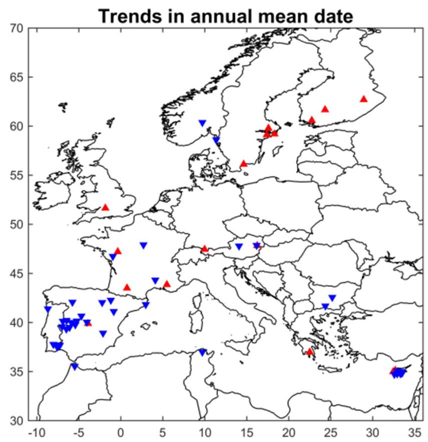

Figure 2.8 Significant increasing (later date, red triangle up) or decreasing (earlier date, blue triangle down) trends in the mean date of zero-flow day occurence, at the 10% significant level (Tramblay et al., in review)

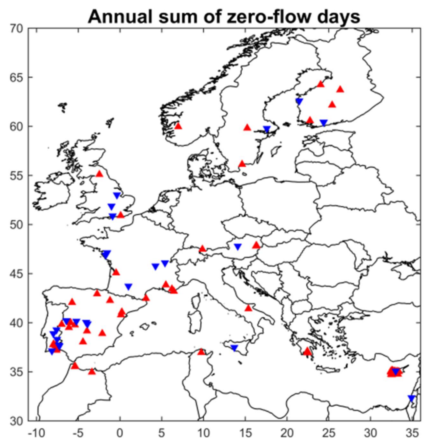

Figure 2.9 Significant increasing (red triangle up) or decreasing (blue triangle down), at the 10% significant level for the mean annual number of zero-flow days (Tramblay et al., in review)

To complement the trend analyses, impact studies on the effect of climate on water resources may support stakeholder decision making. There are few impact studies on European temporary streams due to the difficulties for a hydrological model to simulate flow intermittence and by the reduced number of gauged basins. De Girolamo et al. (2017) have calibrated and applied the SWAT model forced by three regional climate projections to a temporary river system in southern Italy. The results indicated a longer period with no-flow conditions during the period 2030–2059 compared to the recent past period (1980–2009). Beaufort et al. (2020) have quantified the changes in river flow intermittence across France over the 21st century using the modelling framework developed by Beaufort et al. (2018). An ensemble of 26 regional projections derived from GCM simulations under RCP2.6 and RCP8.5 emission scenarios has been used as inputs. Results for the two 30-year periods 2021-2050 and 2071-2100 show an increase in regional probability of drying of headwater streams (RPoD) with time (Figure 2.10). The mean RPoD over the whole period May–October is 11% at the national scale under the current climate, compared to 16% and 20%

on average all RCPs together for the periods 2021-2050 and 2071-2100, respectively. Using a global hydrological model is uncertain when assessing changes in flow intermittence characteristics of temporary streams located in headwaters (e.g. Döll and Schmied, 2012). Globally, in the future, flow intermittence in summer should increase where climate is projected to be drier, conversely flow intermittence in winter should reduce where zero-flow conditions are due to freezing.

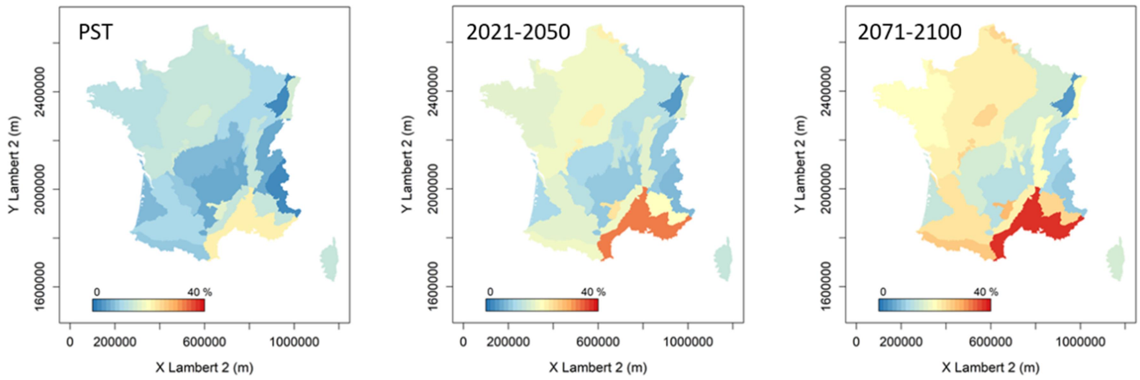

Figure 2.10 Ensemble mean of regional probability of drying of headwater streams for two 21st century time slices in France, reproduced from Beaufort et al. (2018)

2.5 Influence of human activities of the hydrology and morphology of IRES

Accurately describing and analysing the regime of IRES is of great importance to properly preserve or restore ecosystems in these systems. As for perennial regime, the flow regime reflects climate and catchment structures (topography, geology, land cover, human activities). Dry phases in IRES are frequently exacerbated by human activities and can impair most riverine ecological processes. Conversely, the release of water from dams or from sewage treatment plants and the construction of dams can reduce dry phases and also affect biological community structures. Highlighting natural controls on the flow regime of IRES and their respective influence compared to human influences is crucial to determine what measures are needed to manage water uses and biological integrity.

Alterations to flow regimes by human activities threaten the ecological integrity of river ecosystems and may result in serious declines in biodiversity and the provision of crucial ecosystem services. This “natural flow regime paradigm” proposed in a seminal paper by Poff et al. (1997), is why river ecologists and water resource managers rely on detailed information about flow regimes: flow regime fundamentals.

2.5.1 Assessing and characterizing hydrologic alteration in IRES

To assess the degree of hydrologic alteration in fluvial water bodies, following the recommendations of the Ecological Limits of Hydrologic Alteration (ELOHA) framework (Poff et al., 2010), it is necessary to:

i. Find the ‘baseline’ or reference unimpacted regime characteristics for the water body under study,
ii. classify the stream regime using ecologically relevant variables,
iii. determine the deviation of the current regime from the baseline-condition one and
iv. develop regime alteration-ecological response relationships.

The sub-sections above try to provide some recommendations for the two first steps. As the two last steps are necessary not only for assessing the quality of the actual stream regime but also for designing ecological or environmental flows (Eflows), they are detailed below in the sections 6.3.1 and 6.3.2 respectively.

2.5.2 Assessing hydromorphological alteration in IRES

Hydromorphological alterations arise from the fact that IRES during the dry phase are often not considered as rivers supporting biodiversity (see chapter 5). IRES are thus used for livestock grazing, as paths, operated for leisure activities (hiking or quadbiking) or even dump sites.

As discussed above (section 2.2.2), another issue is the sediment management of IRES is its accumulation in the lowland areas and very often in their river delta. The use of heavy vehicles and other machines for vegetation, debris and sediment removal should be strictly prohibited. Very often the lowland river areas and the river delta experience floods and conflicts arise between the river managers (or the authorities) and the local population on the selection of methods to remove sediment and debris during the dry period.

In a short period of time during the hazardous floods the water fills rapidly the main branch, breaks any existing protection walls and overflow out of embankments resulting in flooding of well-cultivated areas and agricultural settlements. Where restoration actions needed natural processes should be considered and where possible include nature based solutions.

2.6 Take-home messages

The basic definition of an IRES is a stream that ceases to flow, and may dry, but this hides a wide spectrum of hydrological behaviours which support various biogeochemical processes and biological functioning.

- The hydrological behaviours can be described by the flow regime - how the streamflow changes over time at a given point (Poff et al., 1997). Analysis of hydrographs can distinguish wet and dry periods, and metrics on the zero-flow period help to characterize the flow regime of IRES. Important hydrological metrics include: timing, frequency, rate of change and duration of the zero-flow period.

- Reducing the flow regime into the “wet” and “dry” periods excludes some biologically relevant hydrological conditions, such as ponded patches or subsurface flow. So a new approach introducing six aquatic states was defined to describe the flow regime of an IRES. From the wettest to the driest, these aquatic states are: hyperheic, eurheic, oligorheic, arheic, hyporheic and edaphic. They can be obtained from qualitative information and thus from various sources.

- Considering these aquatic states introduces the notion of hydrological connectivity being defined as the different paths taken by water, which is of great importance to understanding biogeochemical and biological processes. The three dimensions of hydrological connectivity should be considered: vertical (interactions between surface and groundwater), lateral (between the channel and the riparian zone and wider environments) and longitudinal (from upstream to downstream).

- Various hydromorphological methods have been developed to fulfil the WFD requirements. Two of them can be fully applied to IRES, River Style and IDRAIM, as the discipline of fluvial morphology is based on the processes which shape channels and floodplains, not the persistence of water. This kind of method becomes irreplaceable for very dry streams where hydrology or aquatic ecology can fail.

- The lack of quantitative information provided by the few gauging stations existing in IRES is partly offset by the development of new tools which provide valuable qualitative information, including citizen science initiatives, field loggers, remote-sensing data and aerial photographs. This qualitative information has the advantage of being able to cover a larger area but records are often discontinuous and only for a short period of time depth.

- Classifications based on hydrological metrics are useful to inform the design of sampling schedules and to assess ecological status. As an example, TREHS classification is based on the three aquatic phases and takes advantage of any information available (see section 2.4.2).

- Hydrodynamic models are used to design Eflows (see chapter 6) and have been tested in various IRES (see section 2.4.3). Hydrological models should be applied with caution when studying fine resolution flow intermittence but they are well suited for a global assessment of water resources. Alternative modelling methods are being developed to address the current limitations of the models by simulating no-flow conditions and will benefit from the addition of qualitative data such as the ones collected by citizen sciences initiatives.

- Intermittence of IRES may be natural but it can also be caused or changed by artificial influences such as water abstractions, channel modification or dam construction.

- Regime metrics and classifications are useful tools for describing habitats and hydrological alteration.

- River basin management authorities should go beyond the use of (absolutely necessary) gauging station networks and begin to develop permanent networks for systematic observations of qualitative hydrological information such as the French ONDE network. Water and biological sampling schedules implemented to meet the WFD requirements should include the collection of the aquatic state of the stream reach at the moment of sampling.

- Research studies are ongoing to assess observed and future trends in IRES. Quantitative assessment of the relative influences of natural intermittence and human pressures remains a challenge.

3. Water Physicochemistry in IRES

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3.1 Introduction

3.1.1 In a nutshell

- Spatial and temporal variability of water physicochemical characteristics is higher in IRES than in perennial rivers and streams.
- IRES hydrological regime has a strong influence on abiotic and biotic processes controlling stream physicochemical conditions.
- First-flush events during the rewetting phase have highest concentrations of nutrients and dissolved organic carbon
- Assessment of IRES ecological status through water physicochemical measures needs water quality standards and monitoring strategies that include hydrological temporal dynamics.

3.1.2 IRES as dynamic physicochemistry systems

Intermittent rivers and ephemeral streams (IRES) are characterized by hydrological regimes with extreme events (floods, droughts). This includes the alternation of wet and dry phases, and highly variable lateral, vertical, and longitudinal hydrological connections between stream surface waters and surrounding terrestrial and groundwater environments (chapter 2). These hydrological regimes exert a strong influence on key abiotic (e.g., dilution, hydrological connectivity) and biotic (e.g. microbial nutrient uptake and mineralization) processes occurring within the stream and its catchment. These processes control stream water physicochemical characteristics (i.e. pH, salinity, dissolved oxygen, nutrient concentrations) and have relevant effects on stream biological communities (chapter 4); and thus on the ecosystem services provided by IRES (chapter 6).

Due to their highly variable hydrological regimes, IRES show acute changes in the magnitude of stream physicochemical parameters, thereby having high spatial and temporal variability (Gómez et al., 2017; von Schiller et al., 2017a). During the drying phase, stream discharge, size of the stream network, and the hydrological connectivity within the catchment are dramatically reduced, ultimately leading to fragmentation and cessation of stream flow. Low discharge is associated with a low dilution capacity of the surface water to the inputs from adjacent compartments (e.g. riparian and hyporheic zones) and an increase in hydrological retention and stream water residence time. Moreover, the loss of hydrological connectivity reduces inputs of solutes from terrestrial ecosystems (at least natural inputs, see section 3.4 below) and increases the relative importance of in-stream processes on water physicochemical characteristics and their spatial variability (Figure 3-1). During the rewetting phase (i.e. first-flush events), hydrological connectivity is restored. This rewetting boosts the mobilization and transport of solutes accumulated in terrestrial soils and streambed sediments during the dry phase and reduces the spatial variability in stream water physicochemical characteristics (Figure 3.1). Although patterns of variation in these parameters may differ considerably among different types of IRES, their high temporal and spatial variability must be considered to properly assess the ecological status of these ecosystems.

Environmental policies, such as the European Water Framework Directive (WFD) (EC, 2000), consider water physicochemical characteristics in the assessment of the ecological status of streams. This includes the characterization of thermal, oxygen and acidity conditions, along with salinity, nutrients, priority pollutants and other specific pollutants (Annex II of Directive 2008/105/EC, later amended by the Directive 2013/39/EU) discharged in significant quantities into streams. Therefore, parameters like temperature, dissolved oxygen, pH, conductivity, the concentration of nutrients, insecticides, herbicides and metals are of special interest when monitoring the physicochemical status of any given stream or river, including IRES.

Strategies for physicochemical monitoring differ among EU members, and, so far, few adaptations to the particularities of IRES have been undertaken (Prat et al., 2014; Sánchez-Montoya et al., 2012). The use of physicochemical parameters for ecological status assessment entails previously established reference conditions that should encompass the full range of values expected to occur naturally in each stream type. Within this context, the European Commission has published a guide of best practices for establishing nutrient concentrations to assist Member States in determining the levels of phosphorus (P) and nitrogen (N) that are likely to support good ecological status (Phillips et al., 2018). This guidance can be used to check existing boundary values or to develop new ones. Nevertheless, the range of values can lead to misleading interpretations when applied to IRES because of the high variability of physicochemical parameters observed naturally in these freshwater ecosystems. Hence, it is fundamental to consider the natural variability of IRES in order to properly manage and monitor water quality of these ecosystems (Prat et al., 2014).

This chapter briefly reviews existing knowledge on the physicochemistry of IRES and how it can be used to monitor and manage these ecosystems. We first describe the temporal and spatial patterns of key physicochemical parameters in IRES. Then we provide recommendations for an effective physicochemical monitoring of these ecosystems. Finally, we highlight critical issues that must be addressed for proper assessment and management of water physicochemical quality in IRES.

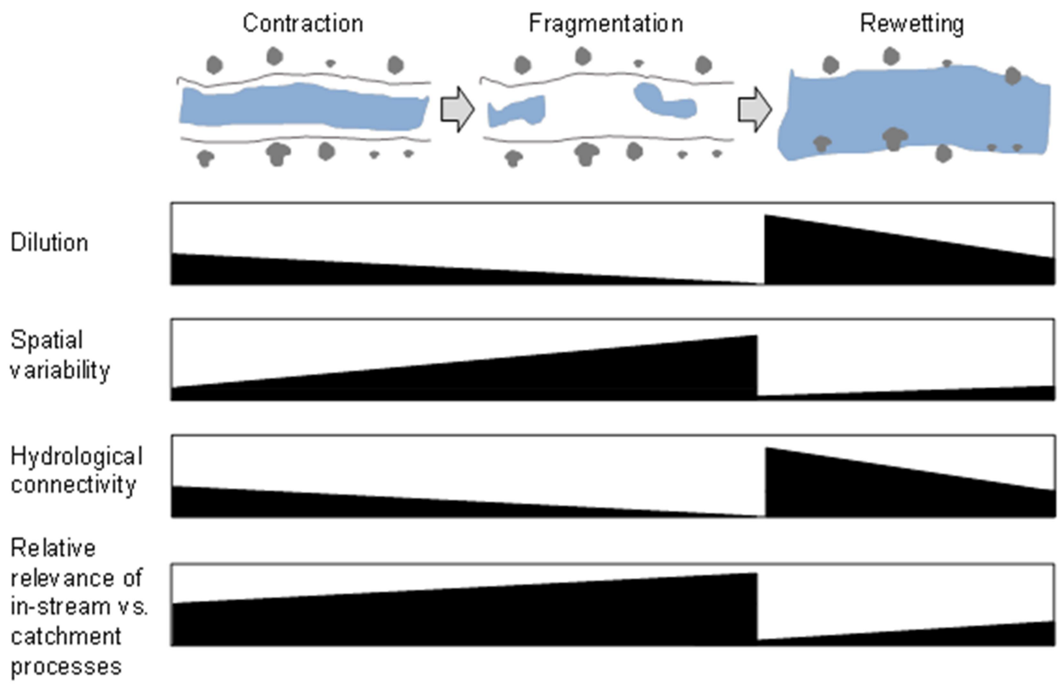

Figure 3.1 Representation of the temporal pattern of key drivers of stream physicochemistry during contraction, fragmentation, and rewetting phases in a typical intermittent stream.

3.2 Temporal and spatial patterns of water physicochemical parameters in IRES

The shift in dominant abiotic and biotic factors that IRES experience through the different hydrological phases (Figure 3.1) results in distinct spatial and temporal patterns of variation in physicochemical parameters. A good understanding of these patterns is important to properly interpret the physicochemical data obtained during sampling and monitoring programs. Here, we describe the general patterns of temporal and spatial variation for key physicochemical variables commonly used in the assessment of the ecological status of streams.

3.2.1 Temperature

Many factors (solar radiation, suspended particles in the water column, depth, water velocity, shading, groundwater inputs, upstream sources, tributaries), influence stream water temperature and its seasonal and daily patterns in surface waters (Brown and Hannah, 2008). The effect of all these factors on water temperature is largely evident in IRES, especially when surface flow discharge diminishes. Under this situation, IRES are highly susceptible to rapid heating and cooling, and thus can show wide diel oscillations in temperature. In general, diurnal variation in water temperature increases as surface discharge decreases along the contraction phase. In those regions where flow cessation coincides with high levels of solar irradiance, water temperatures tend to increase during the drying phase, especially in open canopy reaches. Moreover, fragmentation of surface water and formation of remnant pools increase the spatial variability in mean water temperature and diel oscillations. Pool depth strongly influences temperature patterns. Shallow pools tend to heat up faster than deeper pools. For instance, deep (ca. 4 m) residual pools in ephemeral rivers can thermally stratify for several months (Baldwin and Wallace, 2009).

3.2.2 Dissolved Oxygen

Dissolved oxygen (DO) concentration changes mainly in response to seasonal and daily shifts in photosynthesis, respiration, water turbulence and temperature. In-stream photosynthesis increases DO, while respiration decreases DO. During stream contraction, photosynthesis, and thus DO, can increase in pools and certain reaches, especially at mid-day when water temperature increases in highly irradiated habitats, even leading to DO supersaturation (when DO exceeds 100% saturation at a given temperature) (Boulton and Brock, 1999). On the other hand, DO is consumed by microorganisms during organic matter (OM) decomposition, an important pathway of carbon and nutrient cycling. In this context, large accumulations of naturally- or human-derived OM in sediments can reduce DO concentrations, especially in slow-flowing and drying streams. For instance, DO concentration acutely diminishes in IRES during the fragmentation phase (Figure 4-2) before stream drying, when water temperature and salinity increase (the last by enhanced evapotranspiration) and oxygen solubility decreases (Ylla et al., 2010).

Changes in DO can be highly variable in IRES during the fragmentation phase, depending on the factors that promote either photosynthesis or respiration processes. In addition, increases in water temperature contribute to enhance metabolic activity, and thus DO demand, especially at locations where OM accumulates. In general, in the rewetting phase, turbulence helps DO to increase again (Ylla et al., 2010; Figure 4-2). In ephemeral streams, flow resumption after major rainfall and an increased metabolic activity can rapidly reduce DO, resulting in hypoxia or anoxia in the wetted front (i.e. black waters; Hladyz et al., 2011). Oxygen availability in the water column also can be affected by local groundwater inputs, seepages, or the activity of microorganisms in the hyporheic zone, among other factors. High respiration rates in the hyporheic zone can lead to hypoxic conditions (i.e., DO concentration < 2 mg/L) at the streambed, especially at night. In IRES, it is expected to observe both a wide range in DO concentration, seasonally and daily, as well as an increase in spatial variability as surface flow discharge diminishes.

3.2.3 pH

The pH is considered a key variable because it controls nutrient and metal availability. Values of pH in water and sediments can vary widely due to natural causes, such as catchment lithology, seasonal and daily shifts in photosynthesis and respiration, and variation in humic acid concentrations. Anthropogenic activities, such as acid rain, mining and soil clearing also can change stream water pH. As similarly described for DO variation (section 3.2.2), all hydrological processes in IRES that favour photosynthesis, such as the formation of isolated pools in open canopy habitats, can increase water pH during daytime. However, as stream drying progresses, an increase in respiration rates in pools can cause a decrease in pH (Dahm et al., 2003; Hladyz et al., 2011) (Figure 3.2). Ephemeral streams can also exhibit a decrease in pH after flow resumption in response to high loads of OM (mostly leaf litter) that fuels microbial respiration.

3.2.4 Salinity

Saline rivers and streams occur in catchments with the presence of evaporitic rocks from the Miocene or Triassic, and are found worldwide. The presence of salts and the occurrence of IRES usually are geographically linked, especially in arid regions. However, not all IRES are naturally saline, and catchment lithology is an important factor to consider when interpreting high values and spatial variability of salinity in IRES.

Increased evapotranspiration during low flow and fragmentation periods results in the concentration of salts and potentially acute values of salinity in surface waters (Figure 3.2). This natural process contributes to an increase in the temporal variability of salinity in IRES at daily, seasonal, and annual scales. Additionally, groundwater inputs can increase surface water salinity if groundwater has filtered through water-soluble minerals or rocks of marine origin (Herczeg et al., 2001). In addition, the differential chemical composition of groundwater inflows (local, intermediate and regional flows) is especially relevant for understanding the spatial variability of surface water ionic composition. In IRES, increased concentrations of sulphate (SO_4^{2-},) sodium (Na^+), calcium (Ca^{2+}), silica (SiO_2), potassium (K^+) and chloride (Cl^-), have been described in concert with surface flow expansion after storms (Al-Qudah et al., 2015) and snow/ice melt (Robinson et al., 2016) related to the flushing of weathered materials. In contrast, rainfall events often dilute the high levels of dissolved ions in naturally saline IRES; consequently, water salinity and conductivity decrease (Gómez et al., 2017). Finally, water salinity often increases due to non-point (agricultural, urban runoff) and point (urban, industrial effluents) pollution sources.

3.2.5 Nitrogen, phosphorus and dissolved organic carbon

IRES show a unique 'biogeochemical heartbeat' with pulsed temporal and spatial variations in nutrient and organic matter (OM) inputs, in-stream processing, and downstream transport (von Schiller et al., 2017a). The concentrations of nutrients (dissolved N and P) and dissolved organic carbon (DOC) are affected by the interplay between abiotic (e.g., dilution, adsorption, precipitation, dissolution) and biotic (e.g. biological uptake and release, nitrification, denitrification) processes. The extreme hydrological variability in IRES, coupled with typically low surface discharge and thus low dilution capacity, result in a disproportionate effect of abiotic/biotic processes on nutrients and DOC concentrations.

Surface water quality in IRES responds to processes occurring in the stream channel as well as those in subsurface compartments and adjacent terrestrial ecosystems in relation to local hydrological linkages. However, hydrological disconnection during stream fragmentation and drying diminishes lateral connections through runoff and vertical connections through groundwater. Therefore, surface water quality during the drying phase mostly depends on in-stream abiotic and biotic processes occurring at the stream surface (Gómez et al., 2017). Under these conditions, hydrological retention (water residence time) also increases, which can further enhance the extent of such processes. During drying, surface and subsurface flow inputs (natural or anthropogenic) to the stream can be relevant sources of nutrients and DOC, thereby influencing surface water quality (see section 3.4).

During low flow in summer, photosynthesis likely increases in open canopy shallow reaches and pools because of high solar irradiance, and consequently, this can result in a decrease in nutrient concentration as a result of higher biological nutrient uptake demand. On the contrary, if primary producers (aquatic plants and algae) are scarce, nutrient concentrations may increase as a result of increasing water evaporation (Figure 3.1). Similarly, high water temperature stimulates microbial activity raising OM decomposition rates, and thus increasing DOC concentration in surface waters. Lastly, summer low flows in IRES with carbonate rocks and Iron-rich sediments in riverbeds often show low concentrations of dissolved P in surface water.

Hypoxic conditions can increase as stream drying progresses, especially in OM-rich habitats like pools, which typically results in a decrease in surface water nitrate by denitrification (Figure 3.2). However, low redox potential in OM-rich sediments favour the release of P and ammonium from sediments to the water column. The heterogeneous distribution of sediment redox conditions along reaches during the fragmentation phase increases spatial variability in nutrient and DOC concentrations in surface waters.

Processes occurring in dry channels have significant effects on water quality during river rewetting. Rewetting causes leaf litter accumulation on dry riverbeds to increase nitrification in dry sediments, physical disruption of aggregates, and cell lysis by osmotic shock. Further, extracellular enzyme activities disproportionately increase N, P and DOC during flow resumption (Figure 3.2) (Ylla et al., 2010; von Schiller et al., 2017a).

3.2.6 Pollutants

The presence of pollutants in streams is mostly a consequence of non-point (agricultural, urban runoff) and point (urban, industrial effluents) anthropogenic inputs. The temporal pattern of pollutants in rivers usually is linked to the activities of origin. This is true for all streams, but in IRES, because of their typically low dilution capacity, the effect of pollution inputs on chemical surface water quality can be more noticeable.

Most pollutants from agriculture/urban activity enter streams through water and sediment runoff after rainfall events. If inputs occur after a dry period, the low river discharge in IRES leads to an acute increase in the concentration of pollutants in surface waters and sediments. Some organic contaminants (e.g. polycyclic aromatic hydrocarbons - PAHs) have higher affinity for streambed sediments and suspended particulate matter owing to low aqueous solubility and high hydrophobicity. Moreover, pesticides adsorb more on dry sediments than on moist ones (Götz et al., 1998), which may increase their persistence in IRES in contrast to perennial rivers. During the dry period, polluted sediments remain in IRES riverbeds for long periods of time because water surface flow is low and sporadic, and has a low sediment transport capacity. During this period, a decrease in the concentration of organic pollutants can occur due to photolysis (solar induced degradation) as well as to adsorption that occur within the sediment matrix (Zoppini et al., 2014). In contrast, inputs of pollutants from point-sources are relatively constant and their concentration in receiving streams is mostly driven by their dilution capacity, which varies with the hydrological phase (Figure 3.1) (Zoppini et al., 2014). However, we also need to consider the impact of rainfall events when regarding point source inputs, especially sewage overflows, as they can have a dramatic impact on the IRES water quality.

During the fragmentation phase, spatial variation in sediment redox and pH conditions can notably influence metal concentrations and bioavailability in surface waters. Low redox potential and low pH can lead to the release of metals adsorbed in sediments to the water column.

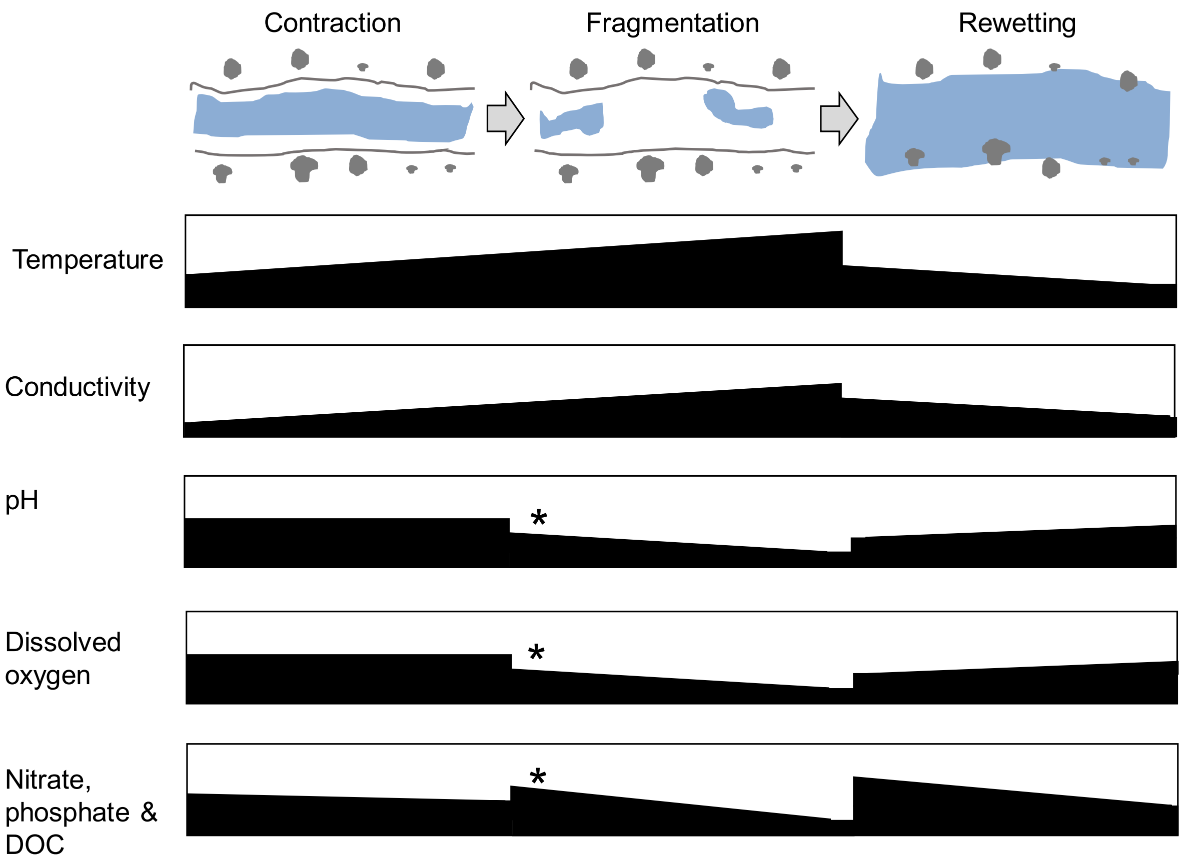

Figure 3.2 Temporal patterns of key physicochemical parameters over different hydrological phases. * indicates that changes occurring under specific conditions of low solar irradiation in close canopy reaches; the opposite pattern can be expected under highly irradiated conditions causing high photosynthesis in open canopy reaches.

3.3 Monitoring of water physicochemical parameters in IRES

The high spatial and temporal variability in physicochemical parameters in IRES complicates the monitoring of water quality. There are several issues that water managers must consider regarding when and where to conduct field physicochemical surveys to monitor IRES, which we address in this section.

3.3.1 When to sample?

Selection of the best time and frequency for sampling will depend on the specific objective of the stream survey. In the context of the WFD, Member States may tailor their monitoring sampling frequency according to the conditions and variability of the target stream, which must be justified on the basis of technical knowledge and expert judgment of the selected IRES (Guidance, W. C. E., 2009). In this context, previous knowledge of the hydrological regime of the selected streams is pivotal for the proper design of the monitoring plan; this being especially important in the case of IRES. The requirement of monthly sampling from conventional programs may not be suitable for IRES because they may be dry for several months each year or even for more than a year. If the objective is to assess water physicochemistry quality over the course of a year, sampling during the period of surface base flow may be the most appropriate strategy. In contrast, if the objective is to capture the range of natural (or human-affected) variation in physicochemical conditions, then a more intensive river survey over time is suggested, including sampling at different hydrological phases. Daily variation in water physicochemistry condition is another aspect to be considered when monitoring IRES. If daily variation is a target objective, at least two collections per day should be done, one before sunrise and one at midday. Similarly, collections at the beginning (before stream drying up) and end of the contraction phase would be necessary to catch the effect of the contraction phase on surface water quality. Sampling of the rewetting phase is more complex because of its unpredictability; however, specific water and sediment sampling devices can be implemented for this objective (Obermann et al., 2009).

To detect anthropogenic pressures in IRES (e.g. point-source pollution), a survey during the period of decreasing discharge (i.e. contraction phase) is preferable because the effect of such inputs on water physicochemistry is most evident. Sediment sampling for the assessment of priority substances and other specific pollutants is recommended during drying (Zoppini et al., 2014). Surveys of water and suspended sediments, which act as reservoirs for lipophilic hazardous substances, in IRES after rainfall storms provide valuable information on the impact of discrete and episodic inputs into rivers.

Knowledge of the hydrological variability of selected IRES also facilitates the correct interpretation of monitoring survey results. For example, physicochemical data obtained in a river after rewetting will be related to the duration and environmental conditions of its previous dry phase. In any case, data arising from traditional sampling programs and from more targeted samples must be analyzed in an appropriate manner for the correct interpretation of results (Guidance, W. C. E., 2009).

3.3.2 Where to sample?

Selection of the most appropriate sampling locations will depend on the hydrological period when monitoring is performed because spatial variability increases in IRES as flow discharge diminishes. Surveys during flow fragmentation are complex to approach since different hydrological situations occur simultaneously at different locations. A representative sampling during this phase should include water samples from different existing habitats, such as flowing and stagnant water stretches, and diverse and heterogeneous pools. For proper interpretation of the physicochemical data collected, specific information of additional environmental variables should also be considered (e.g. temperature, depth of pools, presence and amount of coarse OM, water colour, etc.). After stream rewetting, criteria for the location of sampling stations in IRES do not differ from those in perennial streams during base flow. We highlight that physicochemical quality standards derived from the study of perennial streams are difficult to apply to IRES because of the natural hydrological variability and resulting physical, chemical and biotic processes that ultimately influence IRES water quality. Consequently, specific physicochemical quality standards must be established in IRES to successfully monitor ecological status based on physicochemical characteristics.

3.4 Critical issues related to water physicochemical quality in IRES

In IRES, physicochemical changes associated with rewetting events can be considered a critical issue for water management in specific cases. This phenomenon, associated with the unique natural hydrologic regime of IRES can result in an abrupt increase in organic matter, nutrients, and pollutants transported in water and sediments. Increases in these measures could be interpreted as having negative effects when considering water quality standards from perennial streams. However, IRES are naturally subjected to these conditions, and thus they may not be as negative as expected if precautionary measurements are implemented.

Inputs from human activities within catchments have altered concentrations of solutes in streams as well as their temporal regime. This activity has caused critical physicochemical water quality problems in fluvial ecosystems worldwide. However, the effects are more remarkable in IRES due to their low dilution capacity, especially during periods of low surface discharge prior to drying. In addition, increases in solutes during rewetting can be exacerbated in human-impacted catchments because solute concentrations in runoff tend to be higher.

The effects of non-point source pollution from agricultural activities on IRES are more remarkable in irrigated lands during the contraction phase. Under these conditions, low stream flow coincides with the highest inputs of drainage water from irrigated agricultural soils, especially in arid regions. This modifies both the hydrological regime as well as the physicochemical characteristics of IRES. This is an essential issue in naturally saline streams from the Iberian Southeast because the increase of freshwater inputs from agriculture, among other effects, decreases the natural water salinity affecting biological communities (Millán et al., 2011).

Despite technological improvements in wastewater treatment plants (WWTP), effluents from WWTPs still constitute a relevant point source input to surface waters in Europe (EEA, 2018) and elsewhere. Urban and industrial discharges through WWTP effluents (depending on different types and treatment levels) can increase OM and nutrient concentrations in receiving streams, and even introduce hazardous substances, including emerging pollutants and pathogens. These inputs generate abrupt physical and chemical discontinuities along the fluvial continuum, which alter water quality as well as ecosystem structure and function. There is evidence that these changes affect hydrologic and nutrient concentration regimes as well as biotic communities. This knowledge is of critical relevance for IRES management because it provides insights on integrative structural and functional properties of these ecosystems.

Inputs from WWTPs are relatively constant over time. Therefore, the relative contribution of WWTP inputs to stream discharge varies widely in IRES due to variation in flow (Figure 3.3). In fact, there are periods of the year, especially under conditions that drive the drying phase, when the WWTP inflow accounts for 100% of the stream flow (Martí et al., 2010). Under these conditions, streams receiving point source inputs may turn into islands of permanent flow within a highly intermittent fluvial network. This input modifies the patterns of natural spatial and temporal variability in IRES physicochemistry locally, at the reach scale, and at the catchment scale. Within this context, it is important to consider an integrated management perspective that takes into account both WWTP operations and the characteristics of the receiving streams to preserve the ecological integrity of these aquatic ecosystems while a balance is met with societal demand for high quality water resources.

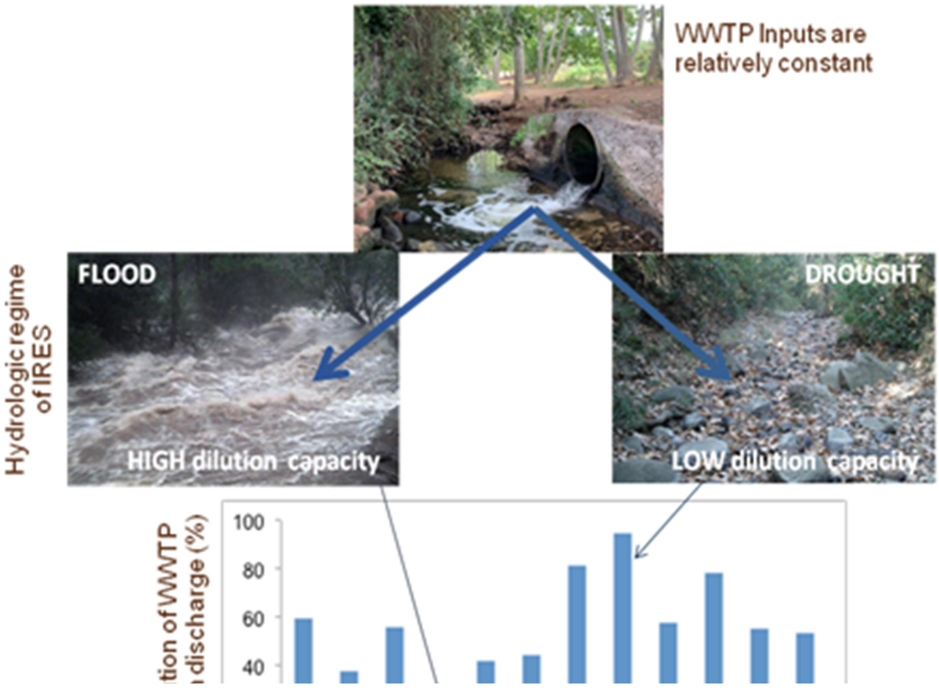

Figure 3.3 Temporal variation in the relative contribution of wastewater treatment plant (WWTP) effluent to discharge of a recipient IRES stream as a result of its hydrological regime.

Inputs of nutrient loads from catchments to streams can be reduced through multiple strategies tracking and managing the source origin, but this is generally constrained by social, political, and economic issues. Therefore, we should seek for additional solutions that may help mitigate the effects of solute inputs into receiving IRES, but also the potential effects of rewetting on downstream aquatic ecosystems in IRES.

Streams have a high bioreactive capacity to retain and transform solutes and nutrients, which can mitigate effects associated with rewetting events as well as those from human activities. However, streams receiving inputs from WWTP effluents tend to show lower nutrient uptake capacity than their stream counterparts with lower nutrient concentrations, mainly because in-stream nutrient uptake becomes saturated as nutrient concentrations increase (Martí et al., 2004). In-stream nutrient uptake and transformation can also be influenced by reach-scale hydromorphological characteristics (Elosegi et al., 2010) by affecting water residence time, habitat configuration, substrate heterogeneity and biological assemblages, which are pivotal factors for nutrient uptake. We expect that reaches with higher water residence time will have higher nutrient uptake because there is a higher interaction between nutrient availability and biological demand. Thus, restoration practices aimed at modifying the habitat configuration of the stream channel, and hydrological linkages between streams and adjacent riparian zones and streambed sediments, can enhance the nutrient removal capacity of streams. Implementation of these practices can be especially useful in IRES because channels have been traditionally impacted by humans (i.e. dumping garbage, crossing roads, channelization). In this context, including Nature Based Solutions in restoration projects that make use of bioengineering techniques, could enhance the effectiveness of in-stream uptake of nutrients, since these techniques use biological components to modify habitat configuration. For instance, helophytes used in bioengineering restoration and their rhizosphere can favour uptake and transformation of dissolved N along subsurface flowpaths.

3.5. Future directions in IRES physicochemistry monitoring and management beyond the WFD

Conservation of freshwater bodies is of paramount importance to prevent the water crisis of this century. The WFD and other water policies should improve and expand their defined boundaries to include IRES (Hoerling et al., 2011). Ideally, the control of IRES hydrology by remote systems would be a key tool for proper monitoring strategies to allow quantifying the spatio-temporal variation during different hydrological phases. In this context, technological applications oriented for assessing freshwater ecosystems and their conservation should be listed in priority international project programs. In addition to the specific mapping and classification of IRES based on their hydrology, water policies should incorporate knowledge on IRES functioning (Stubbington et al., 2018). For instance, the preservation of ecological functioning of IRES by ensuring or restoring hydrological connectivity (longitudinal, lateral, vertical) with other subsystems, and maintaining natural flow regimes, can contribute to ameliorate water quality in IRES. In addition, sediment characteristics and functional indicators associated with dry and wet phases can be relevant characteristics to better assess IRES ecological status (Leigh et al., 2016).

The WFD has promoted the homogenisation of monitoring methods across Europe in perennial streams, precipitating a fundamental reorientation in management objectives from pollution control to ecosystem preservation (Hering et al., 2006). In this context, the implementation of functional indicators in IRES could be of special interest to detect in a fast and complementary way the initial symptoms of water quality deterioration (von Schiller et al., 2017b). Microbiota associated with sediments could be considered as a natural bio-indicator of ecosystem functioning because it responds quickly to changes in hydrological and environmental conditions. Therefore, the study of their activity (e.g., through metabolic assays or with assessment of extracellular enzyme activities) in stream water, remnant pools or even dry sediment, could reveal information about the previous stream hydrological history (i.e. if a stream lost water for a long time or not), as well as about the dominant biogeochemical process associated with a given hydrological phase (Romaní et al., 2013). Recent studies report the importance of physical characteristics of stream sediment in the preservation of microbial ecosystem functions under a persisting dry phase (Gionchetta et al., 2019). For instance, sediments with high capacity to retain water and organic matter would better preserve microbial processes that may influence nutrient cycling upon rewetting (Muñoz et al., 2018).

Another aspect related to sediment monitoring is the analysis of nutrient content. Nutrient concentrations in surface sediments can be seen as the integrated result of short-term processes that occurred during contraction and fragmentation, and could likely be used as a surrogate of stream trophic state. The amount and forms of inorganic nutrients in streambed sediments also represent a potential internal load that can be partially released during flow resumption. Water extractable N and P in dry sediments could help quantify this pool (Shumilova et al., 2019).

3.6. Take-home messages

In this section, we provide a set of take-home messages to inform effective interpretation and monitoring of water physicochemistry in IRES. We also provide some management directions and indicate major knowledge gaps that represent priorities for future research and development.

- The magnitude and range of values for temperature, DO, pH, salinity, and concentrations of N, P and DOC are highly variable in IRES in response to the shift in dominant abiotic and biotic factors that these streams experience through the different hydrological phases (see Figure 3.1; Section 3.2.).

- Remarkable daily variation in physicochemical parameters is a common feature of IRES, especially as streams transition from the contraction to the drying phases. During this transition, spatial variation also increases (see Section 3.2).
- Biogeochemical processes occurring in dry channels have significant effects on water quality upon rewetting, which may constitute a critical issue for water management in specific cases (see Sections 3.2 and 3.4). IRES are naturally subjected to these conditions; thus, precautionary measurements have to be implemented to mitigate their effects.
- Point and non-point source pollution from human activities within catchments have remarkable effects in IRES, especially during periods of low surface discharge prior to drying. Therefore, the concentration of pollutants in surface waters and sediments is commonly more noticeable in IRES than in perennial streams because of their typically low dilution capacity (see Section 3.2).
- Previous knowledge on the hydrological regime of IRES is pivotal for the proper design of water quality monitoring plans and for the correct interpretation of results from monitoring surveys. Selection of time and frequency for sampling depends on the specific objective of the stream survey. Similarly, selection of representative sampling locations depends on the hydrological phase when sampling is performed (see Section 3.3).
- Restoration practices in impacted IRES aimed at modifying the stream habitat configuration and the hydrological linkages with adjacent riparian zones and streambed sediments can significantly enhance their nutrient removal capacity and mitigate effects associated with rewetting events or with human activities (see Section 3.4).
- Published guides for establishing nutrient concentration targets to achieve good ecological status can lead to misleading interpretations when applied to IRES because of the high variability of physicochemical parameters observed naturally in these freshwater ecosystems. To successfully monitor the ecological status of IRES, specific physicochemical quality standards must be established.
- Consideration of sediment characteristics and functional indicators associated with dry and wet phases can be relevant tools to additionally improve the assessment of the ecological status of IRES (see Section 3.5)

4. Community Ecology and Biomonitoring in IRES

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4.1 Introduction

4.1.1 In a nutshell

- IRES instream habitats comprise flowing, ponded and dry patches that shift in space and time to support biodiverse communities of aquatic, semiaquatic and terrestrial species that have the potential to act as biomonitors of ecological quality.
- IRES ecological quality is not assessed in many regions, and where done, biomonitoring typically relies on methods developed for perennial systems. These methods may or may not accurately estimate IRES quality – and in many cases, their accuracy is unknown.
- Best practice involves the use of evaluated or specifically designed biotic indices that consider the aquatic macroinvertebrate communities present during IRES wet phases. In contrast, terrestrial communities remain unexplored as biomonitors of dry-phase quality.
- Future ecological quality assessments may be improved by functional metrics (which explore species traits, not their names), development of genetic tools, recognition of metacommunity dynamics, and by encompassing both aquatic and terrestrial biotas.

4.1.2 IRES as dynamic habitats that support high biodiversity

IRES are typically defined as river ecosystems that sometimes cease to flow and/or dry (Leigh et al., 2016; Datry et al., 2017a) but they can also be viewed as channelized ecosystems that shift between flowing, ponded and dry states (Stubbington et al., 2017a). IRES thus comprise mosaics of wet and dry habitats that vary in space and over time to support ever-changing biotic communities, with lotic, lentic and terrestrial species dominating during flowing, ponded and dry phases, respectively (Figure 4.1; Datry et al., 2014a). IRES communities include vertebrates, notably fish; invertebrates and plants with environmental preferences from fully aquatic to terrestrial; and a diverse range of microorganisms, including diatoms, other algae and bacteria.

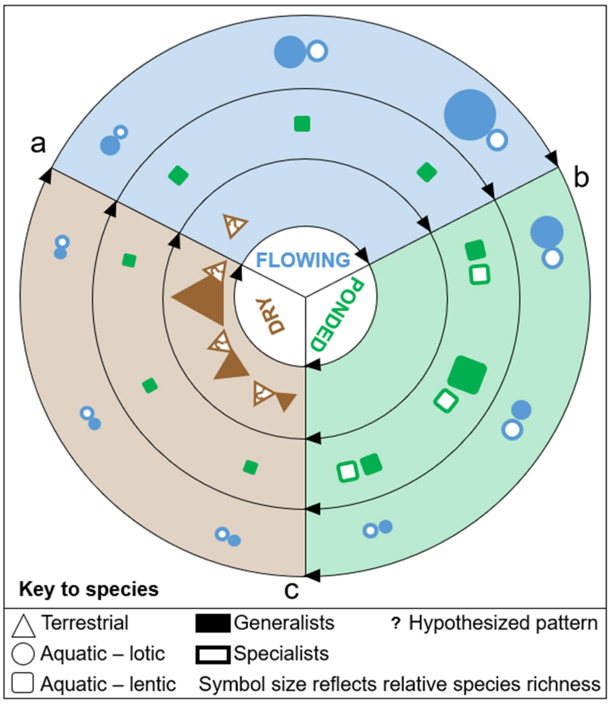

Figure 4.1 The occurrence of lotic, lentic, and terrestrial generalist and specialist species in IRES (a) flowing, (b) ponded and (c) dry phases during a typical sequence of instream habitat changes. Patterns reflect evidence for invertebrates but may also apply t to other groups. *Adapted from Stubbington et al. (2017a).*

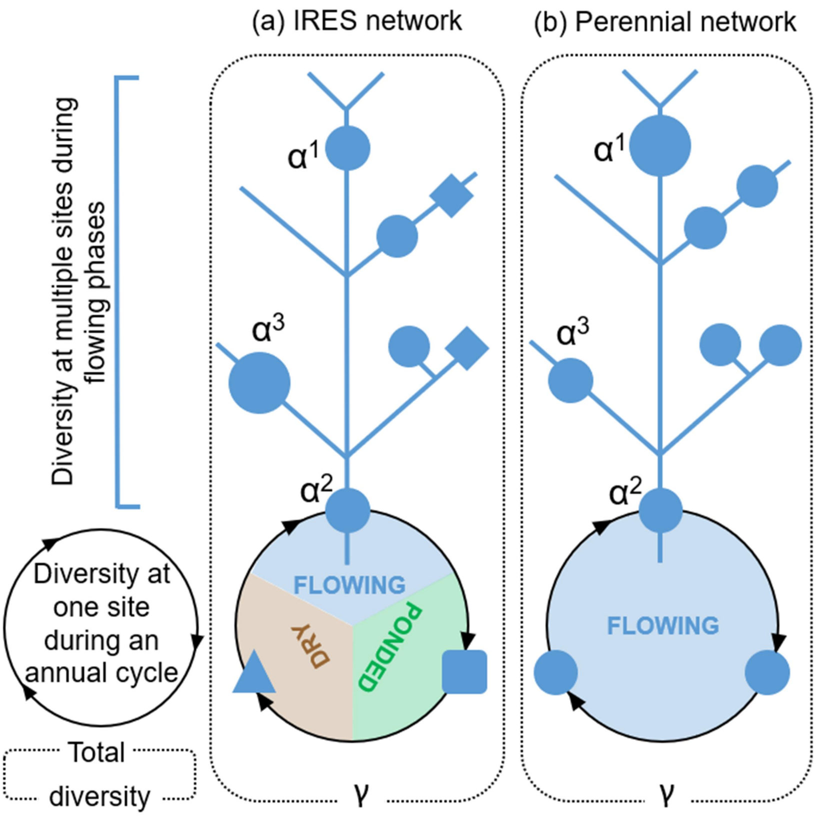

Figure 4.2 Typical α, β and γ community diversity in (a) IRES and (b) perennial networks, at multiple sites during flowing phases (blue lines), and at one site during an annual cycle. Blue symbol sizes equate to diversity and, along with superscript numbers, allow comparison of (a) and (b); symbol sizes should not be compared *within* a pane. Shapes indicate differences in community composition. Patterns are described and definitions given in the text.

Adapted from Stubbington et al. (2017a).

IRES communities can be characterized by their α, β and γ diversities. Alpha diversity describes local richness: the number of species in one place at one time, whereas β diversity refers to variability among sites and times; collectively, α and β diversity determine total, regional-scale γ biodiversity (Figure 4.2; Stubbington et al., 2017a). Compared to those in perennial rivers, IRES aquatic communities typically have lower α diversity, due to the absence of species intolerant of ponded and/or dry conditions (Datry et al., 2014b). These communities may be dominated generalists that also live in perennial streams, but rare and endemic IRES specialists also occur (Figure 4.1; Ferreira et al., 2007a; Armitage and Bass, 2013; Stubbington et al., 2017b); equivalent knowledge of terrestrial communities is lacking. Spatial and temporal β diversity can be higher in IRES than in perennial streams, due to greater abiotic variability (Figure 4.2; Larned et al., 2010). Developing effective strategies to protect biodiverse communities within connected networks of functional ecosystems is a key goal in IRES management (Stubbington et al., 2018a).

In this chapter, we provide information and tools to inform best practice in the ecological monitoring of IRES communities, focusing on European systems managed to meet the requirements of the Water Framework Directive 2000/60/EC (WFD; European Commission, [EC] 2000). We explore fish; aquatic, semi-aquatic and terrestrial invertebrates and plants; and microorganisms, with a focus on well-known diatom communities. For each group, we outline the communities typical of IRES, and explore each group's potential as a biomonitor of ecological quality. A bias towards aquatic invertebrates reflects our current understanding of these communities. We suggest future directions for research informing the development of effective IRES biomonitoring, using molecular tools, trait-based approaches and a metacommunity perspective. Finally, we make recommendations to enable effective decision making by IRES managers seeking to protect biodiversity within these dynamic ecosystems.

4.2 Ecological status assessment in IRES

4.2.1 The legal context for biomonitoring in Europe: the Water Framework Directive

The WFD is the major legal instrument to protect river ecosystems in the European Union (EU). The WFD classifies ecological quality, or status, into five classes, with high, good, moderate, poor and bad classes indicating different degrees of human alteration of monitored 'quality elements' (see section 4.2.2). EU Member States and other participating countries must achieve at least 'good' ecological status (i.e. slight variation from undisturbed conditions) or 'good ecological potential' in 'all' surface water bodies (EC, 2003). However, IRES are not fully recognized by environmental laws including the WFD, in which *temporary rivers* are a classified type only in Mediterranean regions (van de Bund, 2009). As such, existing tools for IRES monitoring, management and conservation are near non-existent compared to those for perennial rivers (Fritz et al., 2017). But despite this limited legal protection, academics, managers and other stakeholders have recognized IRES as valuable ecosystems (Stubbington et al., 2018b) and are collaborating to improve their monitoring and management (Datry et al., 2017b).

4.2.2 Water Framework Directive ‘quality elements’

Ecological status is an expression of the quality of the structure and functioning of surface water ecosystems as indicated by the condition of 'quality elements'. The WFD requires ecological status assessments to encompass several such elements:

- Biological Quality Elements (BQEs, described below);
- Chemical and Physicochemical Quality Elements supporting BQEs, including general elements such as thermal, oxygenation and nutrient conditions; and ‘specific pollutants’, including defined ‘priority substances’ and those released in significant quantities;
- Hydromorphological Quality Elements supporting BQEs, including the quantity and dynamics of flow.

The BQEs for rivers are macrophytes and phytobenthos, benthic (i.e. aquatic) invertebrates, fish, and phytoplankton. BQEs are compared with natural, unimpacted ‘reference conditions’ to assess if biological communities are impacted by human activities. BQE-based objectives drive ecological status assessment, with others described as ‘supporting’ quality elements: abiotic parameters that, at certain levels, support healthy communities (EC, 2000).

4.2.3 Effective IRES status assessment in Europe: current practice and challenges

Across much of Europe, poor recognition has left IRES excluded from or underrepresented in biomonitoring programmes, preventing identification of degraded ecosystems that require restoration or management actions. The Mediterranean regions in which they dominate the network length are something of an exception; here, some IRES status assessments are routinely conducted, although networks are still limited (e.g. Sánchez-Montoya et al., 2010; Nikolaidis et al., 2013; Mazor et al., 2014). However, poor understanding of their ecology and the lack of IRES-specific methods and indices may limit the accuracy with which such biomonitoring can characterize IRES ecological status (Stubbington et al., 2018a).

Most WFD biomonitoring methods have been developed for perennial rivers, including the indices calculated to summarize community health by comparison with reference conditions. Such community-based indices may decline with increasing flow intermittence (e.g. Morais et al., 2004) and thus may not accurately reflect IRES ecological status, as explained below for fish (see section 4.3) and aquatic invertebrates (see section 4.4.2). Other limitations of current approaches include: the WFD ‘water body’ definition, which inadequately represents IRES; difficulties in defining river typologies and reference conditions for IRES characterized by variability; and shorter, more unpredictable opportunities to sample aquatic biomonitors in IRES, as described in Figure 4.3 (Stubbington et al., 2018a).

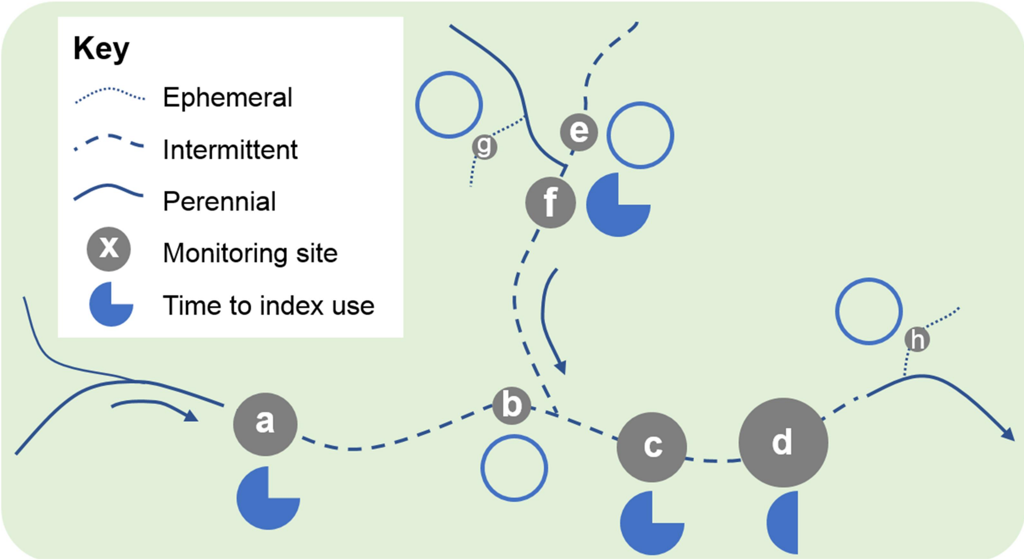

Figure 4.3 Plan view of a river network, indicating the suitability of a perennial biotic index for use in IRES. Size of symbols *a* to *h* is proportional to index 'suitability', which is based on similarity in community composition between perennial sites and IRES during periods of peak biodiversity. Fill of partial circles indicates the period of flow needed before the index is suitable: in IRES with seasonal, predictable intermittence, site *d* and sites *a*, *c* and *f* require 6 and 9 months of continuous flow, respectively, before index use is valid; and at sites *b* and *e*, differences in community composition persist throughout the year, making the perennial index unsuitable. Perennial indices are unlikely to be suitable at ephemeral sites *g* and *h*, where flowing phases are unpredictable and often short.

From Stubbington et al. (2018a) https://creativecommons.org/licenses/by-nc-nd/4.0/

4.2.4 Towards improved IRES biomonitoring and management

Past projects seeking to improve ecological status assessment in IRES include the MIRAGE (Mediterranean Intermittent River Management) Project, which recommended appropriate indices based on aquatic macroinvertebrate community structure (Prat et al., 2014). Building on MIRAGE, the LIFE+ TRivers project created tools including 'TREHS' software (see section 2.4.2) to support biomonitoring done to achieve WFD targets in Mediterranean IRES. The 2016-20 EU COST Action 'SMIRES' seeks to advance IRES management, in part by adapting current biomonitoring methods for IRES across Europe (Datry et al., 2017b). Below, members of the SMIRES *Community Ecology and Biomonitoring* working group (http://smires.eu/working-groups/) explore the challenges and opportunities of using different biotic groups as biomonitors of IRES biological quality.

4.3 Fish

4.3.1 An overview of IRES fish communities

IRES fish communities typically have lower richness than those in perennial systems, but can support species with wide environmental preferences, and – in Mediterranean regions – a high proportion of endemics – as well as highly invasive species, notably the mosquitofish *Gambusia holbrooki* (Figure 4.4 ; Ferreira et al., 2007a). Temporal changes in aquatic habitat availability are the primary determinant of the structure of IRES fish communities (Cid et al., 2017). As aquatic habitats contract, fish can be stranded on drying sediments, with few species – and none in Mediterranean or, to our knowledge, any

European regions – adapted to survive drying (Kerezsy et al., 2017). Fish also inhabit isolated pools during otherwise-dry phases, where deteriorating water quality (e.g. high temperatures and low oxygen concentrations), competition for resources and predation threaten their survival. Some pools are temporary, but those that persist until flow resumes may be refuges, with opportunist feeding modes and tolerance of poor water quality promoting fish survival. Pools and other refuges including perennial reaches, allow fish to quickly disperse and recolonize previously dry reaches after flow resumes (Magoulick and Kobza 2003; Marshall et al., 2016).

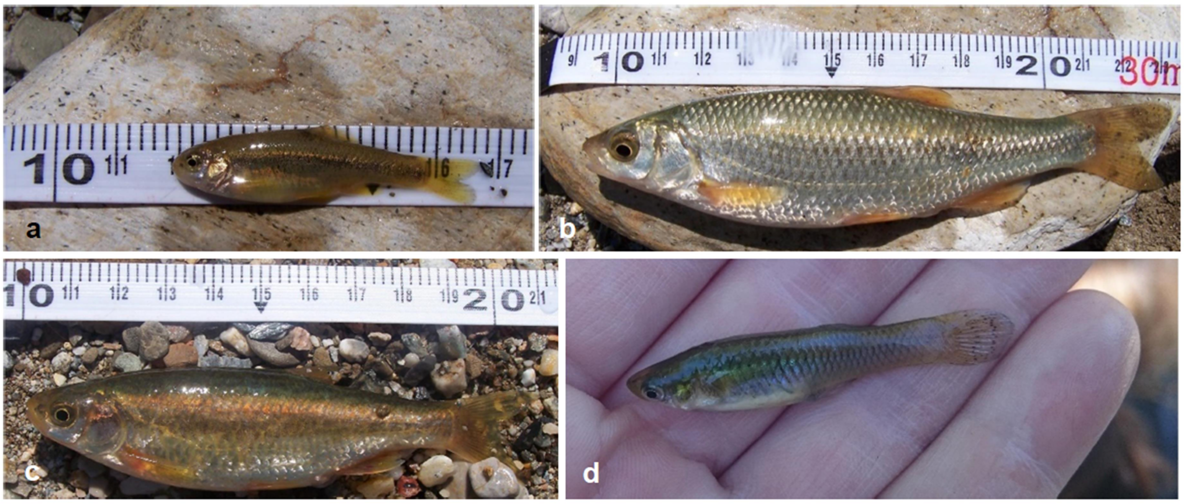

Figure 4.4 Fish species in Mediterranean IRES include the threatened endemic cyprinids (a) *Pelasgus laconicus*, (b) *Squalius keadicus* and (c) *Tropidophoxinellus spartiaticus*, designated as *critically endangered*, *endangered* and *vulnerable* on the IUCN Red List, respectively (Crivelli, 2006a-c), and (d) the highly invasive non-native *Gambusia holbrooki*.

© Louis Vardakas (a-c) and Núria Cid (d).

Fish respond to alteration of hydrological, morphological and (to a lesser extent) water quality variables by human activities (Birk et al., 2012) and are thus potential bioindicators of IRES ecological quality. However, their low α-diversity, the high proportion of endemics, our poor knowledge of their ecology and their wide abiotic preferences all hamper reliable use of fish in IRES biomonitoring (see section 4.3.2; Ferreira et al., 2007a,b). Common metrics used to assess biological quality in perennial rivers, such as species richness, abundance, population density and biomass (FAME Consortium, 2004), may be unreliable in IRES due to temporal changes in community composition. In particular, densities and thus metric scores can peak during the initial stages of habitat contraction (Figure 4.5b), causing overestimation of biological quality using standard indices (Vardakas, 2017). Equally, quality can be underestimated if density-based indices are calculated using data collected after flow has resumed but before communities have recovered from summer mortalities (Figure 4.5c; Noguera Ropero, 2016). Data from surveys conducted during pool phases and following flow resumption should thus be interpreted with caution (Figure 4.5). The effective use of fish as bioindicators in IRES is also hampered by the wide environmental tolerances of many native fish (Magalhães et al., 2002), which limits their ability to detect human impacts.

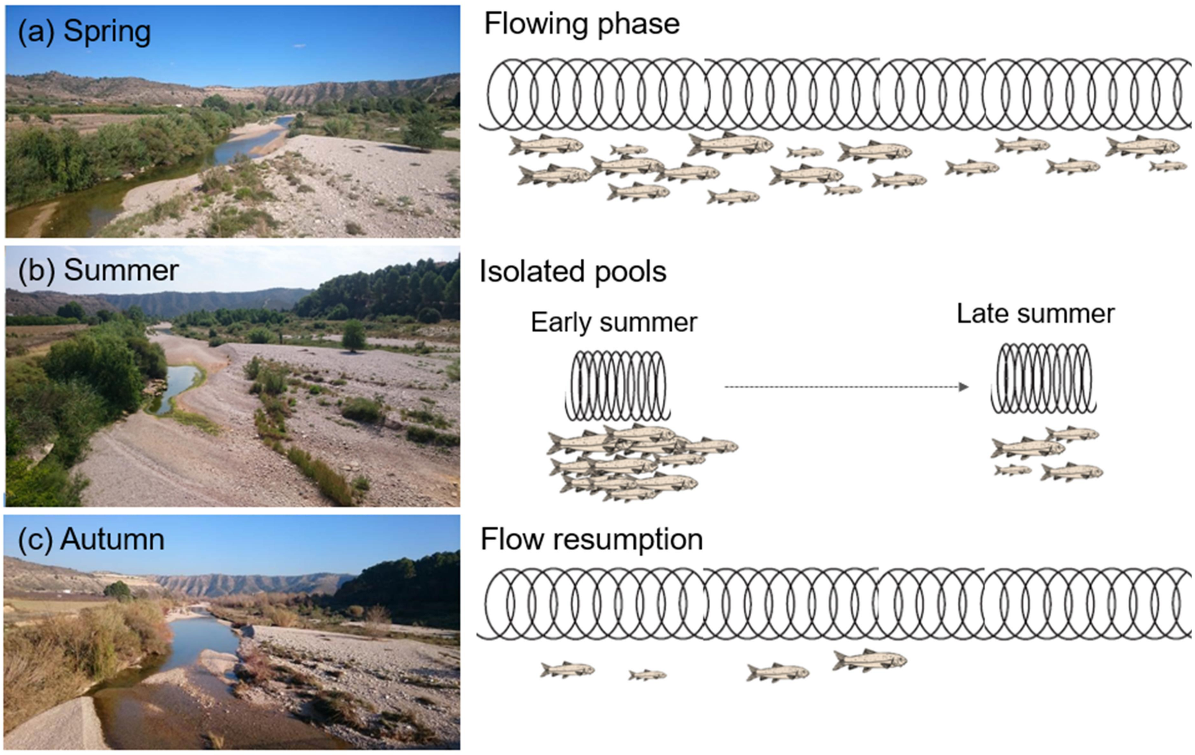

Figure 4.5 Changes in fish densities in relation to seasonal variation in aquatic habitat availability in Mediterranean-climate IRES during (a) spring, (b) early to late summer, and (c) autumn.
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4.3.2 Case study: the development of fish-based indices for Mediterranean IRES

In the context of the WFD, a 'multimetric' fish index (i.e. comprising multiple taxonomic and/or functional response metrics; see section 5.9.3) was developed to assess ecological quality in Mediterranean rivers (Ferreira et al., 2007b). However, the responses of candidate metrics to human impacts were weak and highly variable in IRES due to low local species richness and the prevalence of endemic species, which prevented metric selection. Instead, each country within the Mediterranean ecoregion developed indices effective at either a national scale (Zogaris et al., 2018) or designed specifically for one or few river basins (e.g. Magalhães et al., 2008; Aparicio et al., 2011; Hermoso et al., 2010). IBICAT2b is one such index: a fish-based metric to evaluate the biological status of six (IRES and perennial) river types in the Catalan and Ebro River Basins (de Sostoa et al., 2010), with selected metrics calibrated for each river type. Flow intermittence was not considered in index development, although large river types are classified as 'bad' status if dry, because drying in such rivers indicates artificial intermittence caused by hydrological alteration. For other river types – including all natural IRES – status cannot be estimated using this index if a river is dry, or if insufficient fish are sampled (García-Berthou et al., 2015).

This case study highlights our limited understanding of fish as a WFD BQE in IRES, with further research needed to develop reliable IRES-specific indices that recognize natural temporal variability in their fish assemblages. In the meantime, García-Berthou et al. (2015) and Gallart et al. (2017) provide best practice recommendations aimed at seasonal Mediterranean IRES (for which methods are most advanced), including the adaptation of standard sampling periods to ensure peak diversity is represented.

4.4 Aquatic invertebrates

4.4.1 An overview of IRES aquatic invertebrate communities

IRES aquatic invertebrate communities comprise a diverse range of species that is broadly comparable to that in perennial systems and includes arachnids, crustaceans, flatworms, insects, leeches, molluscs and worms. Insects are often diverse, and include species of beetle, caddisfly, damselfly, dragonfly, mayfly, stonefly, true bug and true fly (Figure 4.6; Stubbington et al., 2017b). Meiofauna (broadly defined as microscopic invertebrates), are abundant community members, but macroinvertebrates (i.e. those visible to the naked eye) are far better studied and are thus the focus of this section. Compared to those in perennial streams, IRES communities typically have lower local species richness than perennial streams (Datry et al., 2014b; Soria et al., 2017), but their assemblages include rare IRES specialists (Armitage and Bass, 2013), and variability among sites enhances IRES contribution to regional diversity (Bogan and Lytle, 2007; Leigh and Datry, 2017). IRES support a higher proportion of drought-adapted species including desiccation-resistant and resilient species which quickly recolonize when flow resumes (Bonada et al., 2007; Cañedo-Argüelles et al., 2016).

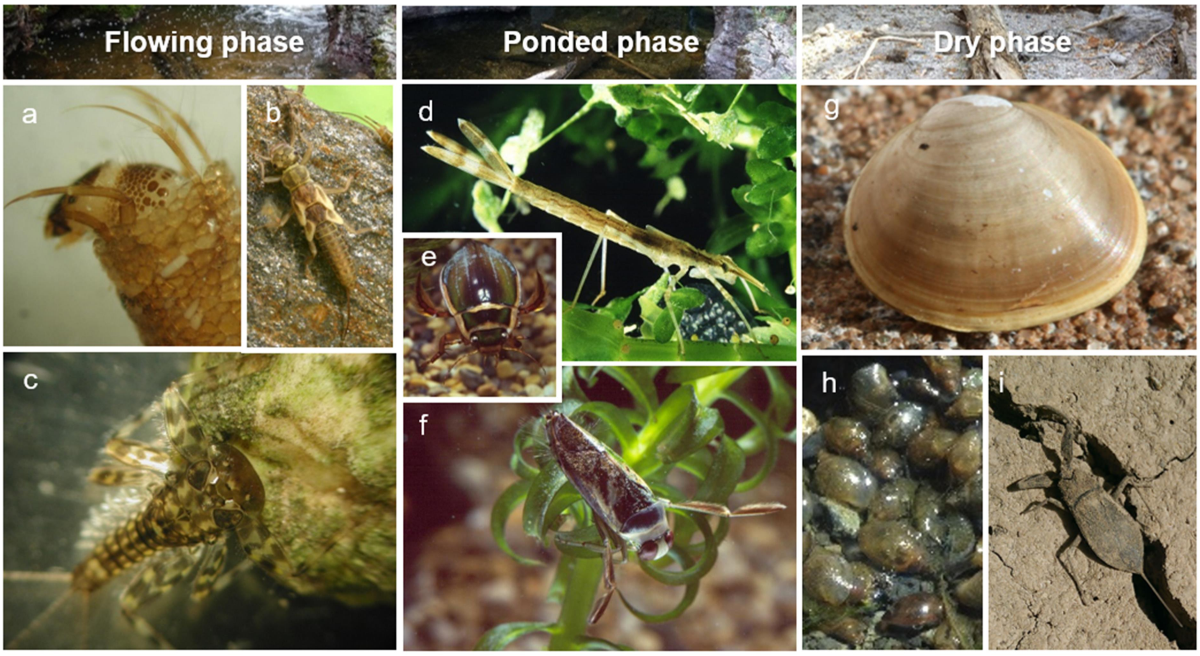

Figure 4.6 Typical IRES aquatic invertebrates include (a) caddisfly, (b) stonefly and (c) mayfly juveniles during flowing phases; (d) damselfly juveniles, (e) beetles and (f) true bugs in ponded waters; and desiccation-tolerant (g) mussels and cased caddisfly larvae, (h) snails and (i) beetles in ‘dry’ habitats.

Adapted from Stubbington et al. (2017b). © Judy England (a, c); Michal Straka (b); Environment Agency (d-f); Rachel Stubbington (i)

IRES aquatic invertebrates are exposed to – and responsive to – a range of human pressures. IRES communities may experience higher pollutant concentrations than those in perennial streams due to reduced dilution and longer persistence times (Chiu et al., 2017). In addition, IRES ‘perennialization’ by artificial water inputs is an IRES-specific stressor and can cause the loss of specialists adapted to flow cessation and/or drying (Stubbington et al., 2017a). Equally, water resource pressures can increase the extent of drying, reducing richness if species-specific tolerance thresholds are exceeded.

IRES biomonitoring is best developed in the Mediterranean Basin, where macroinvertebrate-based methods designed for perennial streams have been adapted. Here and elsewhere, temporal variability in community composition directly drives the selection of IRES biomonitors as well as sampling designs. Family-level richness; Ephemeroptera (mayfly), Plecoptera (stonefly) and Trichoptera (caddisfly; EPT) richness; the Intercalibration Common Metric index (STAR_ICMi; Buffagni et al, 2012); Iberian Mediterranean Multimetric Index (IMMi-T; IMMi-L; Munné and Prat, 2009); Hellenic Evaluation System (HESY-2; Lazaridou et al., 2016); and Portuguese Multimetric Index (IPtIS; IPtIN) are metrics and indices successfully used in Europe to assess IRES biological quality (Prat et al., 2014; Stubbington et al., 2018a; see Table 1). Flowing phases in which connected riffle, ponded and other habitats persist for long enough to be colonized by stable communities may be the most appropriate time to collect biomonitoring samples for analysis using such indices.

4.4.2 The performance of standard invertebrate-based indices in IRES

Benthic invertebrates are among the most common biotic groups (i.e. BQEs, section 4.2.2) used in WFD ecological status assessments in European rivers, primarily by characterization of taxon-specific sensitivities to human stressors such as organic pollution (Birk et al., 2012; Paisley et al., 2014). Most invertebrate-based biomonitoring methods and indices have been developed in, and for, perennial streams. Studies have been done in several European countries, their findings indicating that such approaches may or may not effectively indicate biological quality in IRES. Equally, in many countries, perennial-based indices are used in IRES without their performance having been evaluated (Stubbington et al., 2018a).

Wilding et al. (2018) found that the BMWP (Biological Monitoring Working Party)and its derivative the ASPT (average score per taxon; Armitage et al., 1983) indicated poor biological quality in UK IRES exposed to minimal human impacts, although the ASPT was less affected by drying than BMWP (Figure 4.7).

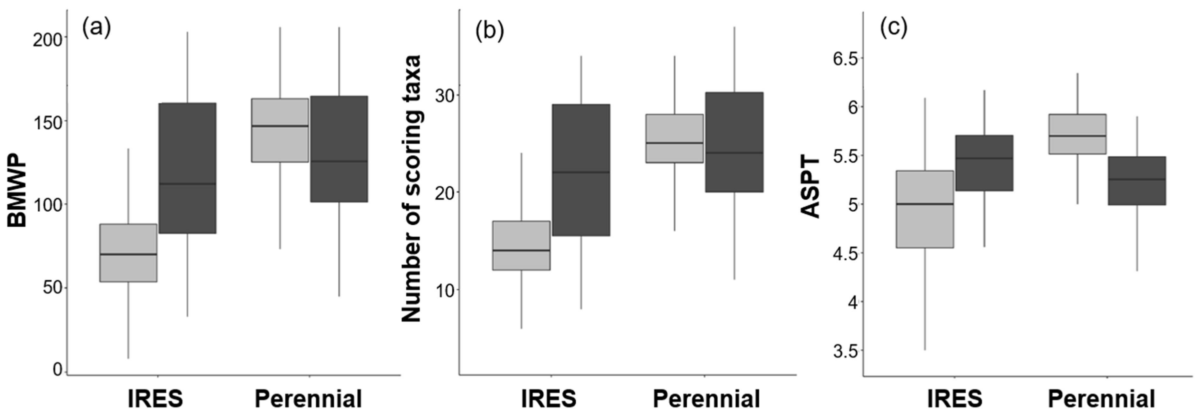

Figure 4.7 Performance of the BMWP index and associated metrics at perennial and IRES sites with seasonal flow regimes on four UK streams: (a) BMWP sample scores; (b) the number of scoring taxa; and (c) the average score per taxon (ASPT). Light grey and dark grey fill indicate streams with different drying patterns; further details are provided by Wilding et al. (2018).

Adapted from Wilding et al. (2018).https://creativecommons.org/licenses/by/4.0/

Similarly, Munné and Prat (2011) showed that the Iberian IBMWP and IASPT (Alba-Tercedor et al., 2002) differed between high and low rainfall periods in unimpacted IRES in Catalonia, suggesting that scores apparently indicative of poorer quality actually reflected drying severity. The poor performance of such indices in IRES partly reflects comparable responses of high-scoring taxa such as mayflies, stoneflies and caddisflies (i.e. EPT, section 4.4.1) to natural low flows, flow cessation and drying, and to human stressors such as organic pollution (Hughes et al., 2009). In contrast, other studies show that single indices *can* reliably indicate IRES biological quality, including the iBMWP in the Catalan region (but only in wet years; Munné and Prat, 2011) and in Croatia (Mihaljević et al., 2011), and the BMWP-type HESY-2 index in Greece (Lazaridou et al., 2018). In such cases, accurate index performance may reflect colonization of ponded waters by beetles, damselflies, dragonflies and true bugs, compensating for the loss of sensitive high-scoring taxa (Bonada et al., 2006).

The biological quality of IRES has also been reliably characterized using invertebrate-based multimetric indices – in which biotic metrics characterizing community sensitivity to individual stressors are numerically combined. Such an approach has been most widely validated in mediterranean-climate countries such as Cyprus, using the STAR_ICMi (Buffagni et al, 2012) and north and south Portugal, with the IPtiN and IPtiS (INAG, 2009). The Iberian Mediterranean Multimetric Indices IMMi-L and IMMi-T have been shown to outperform the iBMWP in IRES across the Catalan region of Spain, and have thus become the indices used by regulatory agencies in the Catalan River Basin District (Munné and Prat, 2009, 2011). However, such multimetric approaches do not always perform well. For example, despite incorporating the preferences of taxa towards temporary waters, the I_2M_2 does not effectively characterize biological quality in French IRES (Mondy et al., 2012; Pelte et al., 2012).

To build on good practice to date and enable the future development and application of biomonitoring indices in IRES, we guide readers to the recommendations made in section 4.10.1. In addition, incorporating understudied groups including meiofauna (see section 4.4.5; Miccoli et al., 2006, 2013) and recognizing species-level environmental requirements of common IRES inhabitants such as segmented worms (oligochaetes) and non-biting midges (Chironomidae; Cañedo-Argüelles et al., 2016) could enhance invertebrate-based index performance in IRES.

4.4.3 DEHLI: an index to characterize invertebrate community responses to drying

The Drought Effect of Habitat Loss on Invertebrates (DEHLI) index describes aquatic invertebrate responses to changes in habitat availability during drought (Chadd et al., 2017). DEHLI operates at family level, with some genus-level adjustments. Each taxon is given a ‘drought intolerance score’ (DIS) based on its association with habitats lost as flow declines, from fast-flowing riffles to isolated pools (Figure 4.8). A sample’s DEHLI score is calculated as the mean DIS for all scoring taxa. Kick sampling is a suitable method in connected surface waters, with modification and /or supplementation by other methods to sample drying habitats (see section 4.5.2). Monthly sampling of set points along a 50 m length is enough to characterize communities while limiting ecological impacts.

DEHLI was designed to assess biotic responses to low flows and partial drying in perennial streams during drought – conditions which also occur as other IRES shift from flowing to ponded and dry states. Sarremejane et al. (2019) tested DEHLI in IRES, and found that it effectively characterized community responses to and recovery from flow cessation and drying at near-perennial sites. Responses were more erratic in more intermittent IRES, perhaps because community reassembly differs among IRES when flow resumes and also because different flow states were sampled. White et al. (2019) identified factors such as habitat structure and drying patterns that could account for these erratic responses and associated spatial and temporal variability in DEHLI scores. The index could be used to describe flow permanence regimes and to identify recent dry phases where flow data are unavailable.

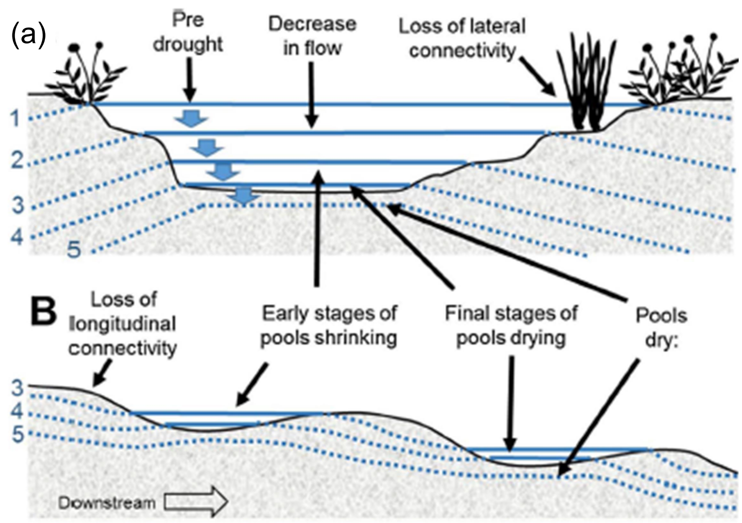

Figure 4.8 (a) A cross-section and (b) longitudinal profile showing sequential changes in hydrological connectivity and wetted habitat as a drought progresses: 1, decrease in flow; 2, loss of lateral connectivity; 3, loss of longitudinal connectivity; 4, contraction of pools; 5, drying of pools.
Adapted from Boulton (2003) and presented in Chadd et al. (2017).

DEHLI was derived from an Australian model (Boulton and Lake, 2008) then adjusted and applied to UK data, indicating its potential for wide application. However, to date, the index has only been tested in groundwater-fed rivers in south England (Chadd et al., 2017; Sarremejane et al., 2019). Application to a broad, Europe-wide range of IRES – including hard-geology and surface-flow systems – will improve understanding of how index scores respond to environmental variability, and could lead to wide adoption of DEHLI as a tool to d (b) ment the ecological effects of drying.

4.4.4 BIODROUGHT: a macroinvertebrate-based index to identify antecedent drying

BIODROUGHT project members have developed a tool that uses benthic macroinvertebrate communities to identify antecedent drying in IRES in Central Europe (Figure 4.9; Straka et al., 2019). To inform tool development, flowing-phase samples were collected in perennial, near-perennial and intermittent streams using a standard 3-minute kick sampling method. Taxa were identified to the lowest resolution possible, typically species. Analyses explored both taxonomic and functional (i.e. trait-based) metrics (see section 4.9.3): the occurrence of indicator taxa, the proportion of various biological traits, and metrics describing richness and abundance. The resultant multimetric 'BIODROUGHT' index indicates the probability that short-term (<7 days) or long-term (7 days to 6 months) drying occurred during the summer before sampling, with different metric combinations used for index calculation based on samples collected in spring and autumn. The index distinguishes perennial streams from IRES with 80-90% accuracy and differentiates short- from long-term drying with lower success rates (Straka et al., 2019).

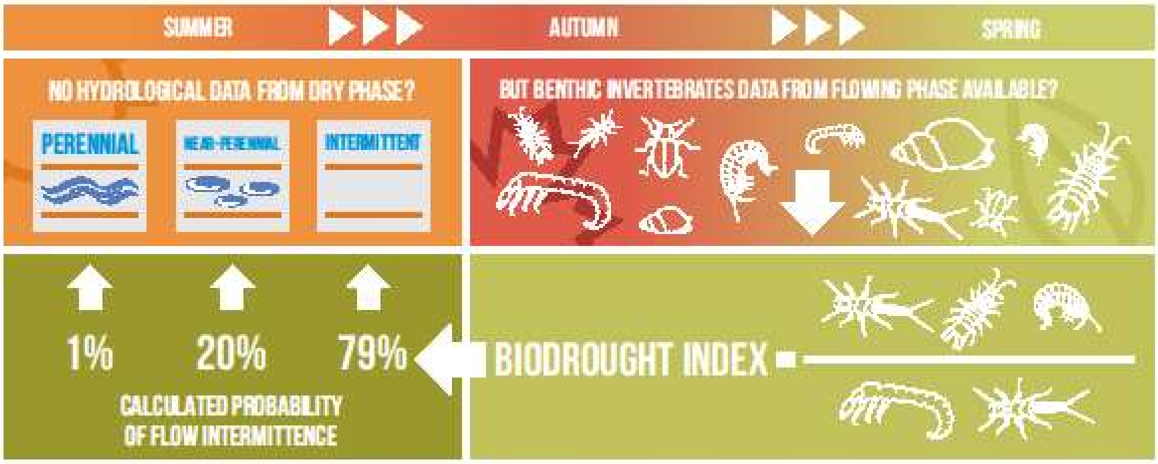

Figure 4.9 Scheme outlining the use of macroinvertebrate sampling data to calculate the probability of antecedent stream drying using the BIODROUGHT index.

From Straka et al. (2019) https://creativecommons.org/licenses/by-nc-nd/4.0/

Given the scarcity of informative data describing IRES flow regimes, we recommend that environmental regulators, water companies and other managers use the BIODROUGHT calculator (http://biodrought.eu/BScalc.php) to identify antecedent drying events. Such knowledge can improve interpretation of routine biomonitoring sample data collected in both perennial streams and IRES. However, the method was developed in unimpacted IRES and its ability to identify responses to drying in impacted systems has yet to be tested.

4.4.5 Mites (Hydrachnidia) as biomonitors of IRES ecological quality

Water mites – the Hydrachnidia – are a diverse and abundant contributor to benthic and hyporheic invertebrate communities in IRES and other freshwater ecosystems, where they are frequently overlooked due to their small size and taxonomic complexity. In IRES, mites can survive dry phases in wet refuges such as isolated pools or nearby perennial reaches, as dormant forms in humid sediments, or by parasitizing emerging freshwater insects. These persistent assemblages remain sensitive to environmental change, and it has long been recognized that their wide range of taxon-specific responses to anthropogenic impacts make mites effective bioindicators (Kolkwitz and Marsson, 1909).

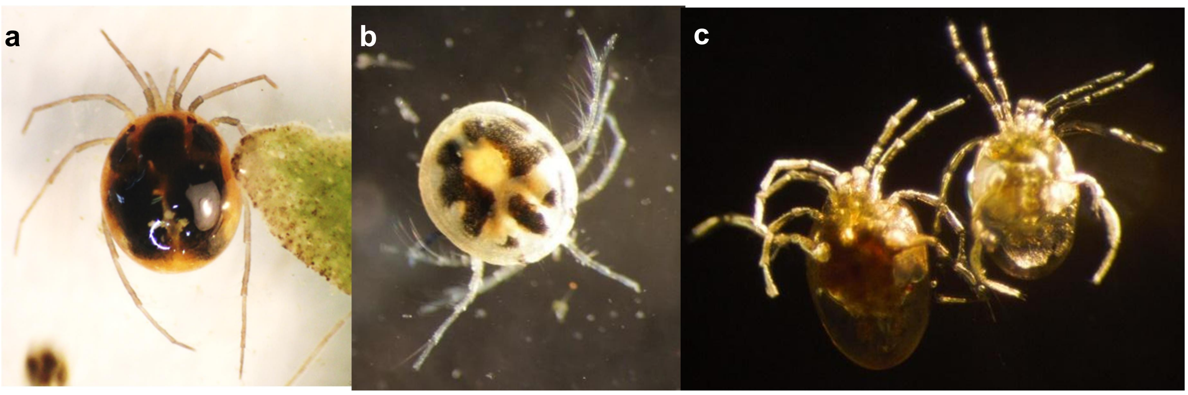

Figure 4.10 The diverse morphology of mites (Hydrachnidia), shown here by the (a) Hygrobatidae, (b) Mideopsidae and (c) Torrenticolidae families, enables their identification and thus potential as IRES biomonitors. Miccoli et al. (2013) show a relationship between family richness and ecological status.

Miccoli et al. (2006) introduced the *PTH* index based on the presence or absence of mites, stoneflies and caddisflies (Hydrachnidia and Plecoptera, Trichoptera) and demonstrated its potential in perennial springs – even without identification beyond the group level. Miccoli et al. (2013) then introduced and tested the *PTHfam* index in other lotic habitats, showing the insights brought by family-level identification (Figure 4.10). Both PTH and PTHfam indices are less responsive to changes in flow permanence than indices based on the abundance and/or richness of EPT (see section 4.4.1), making these indices of potential use in IRES.

To maximize the potential of mites as IRES biomonitors, further research is needed to characterize community responses to different types and levels of human stressors at sites with comparable flow permanence. To maximize their collection in current biomonitoring programmes, surveys should be timed to coincide with seasonal peaks in mite abundance (i.e. spring and early summer). In addition, a smaller mesh size (~200 µm) is recommended to sample mites robustly, although most adults are captured in 500-µm nets.

4.4.6 The aquatic invertebrate seedbank: a potential dry-phase biomonitor

The invertebrate 'seedbank' comprises all life stages of aquatic organisms that remain viable in the IRES sediments during dry phases (Stubbington and Datry, 2013). Seedbanks allow desiccation-tolerant species to persist during dry phases. Assemblages can be diverse, including beetles, crustaceans, caddisflies, mussels, snails and stoneflies, and worms and true fly families including the Chironomidae may dominate. Seedbanks are responsive to abiotic factors including dry-phase duration (Larned et al., 2007), interstitial humidity (Poznańska et al., 2013) and sediment composition (Lencioni and Spitale, 2015). Seedbanks may also differ between unimpacted sites and those affected by human impacts, and assemblages are thus a potential biomonitor of dry-phase biological quality. To investigate this potential, Stubbington et al. (2019a) explored seedbanks from unimpacted sites and sites impacted by sediment mining in a semi-arid region of Bolivia (Datry, unpublished). Species richness and biotic quality index scores were lower at impacted sites, although longer dry phases at impacted sites may also have caused assemblages to differ.

Figure 4.11 Sediment dug from dry IRES can be rehydrated with oxygenated water for 28 days to trigger the development of dormant invertebrates within the 'seedbank'.

© T. Datry.

To characterize seedbank assemblages, replicate sediment samples are dug from the bed then rehydrated with dechlorinated, oxygenated water, typically for 28 days (e.g. Datry et al., 2012). Experiments are thus time-consuming but require minimal infrastructure (Figure 4.11).

4.4.7 Hyporheic invertebrates as IRES biomonitors

The hyporheic zone comprises the subsurface sediments below the streambed, and varies in extent in response to local hydrological conditions and sediment structure (Boulton et al., 2010). After surface water is lost from IRES channels, the hyporheic interstices may remain saturated or humid, or may dry completely (Boulton and Stanley, 1995; Rosario and Resh, 2000). In all streams, hyporheic sediments can support a diverse invertebrate assemblage, including temporary, permanent or accidental inhabitants, and including both meiofauna and macroinvertebrates (Boulton, 2000; Hakenkamp and Palmer 2000). Temporary residents include juvenile life-stages and primarily benthic invertebrates, who migrate into deeper sediments in response to adverse conditions in the surface stream (Stubbington, 2012).

Hyporheic invertebrates are potential biomonitors of biological quality in IRES, because they may persist after surface water is lost. In particular, macroinvertebrate responses to both human impacts and drying are well-characterized, which may enable their use as biomonitors (Leigh et al., 2013). First, however, research is needed to characterize communities in different stream types and how they vary as IRES shift between flowing, ponded and dry states; the next step will be to link to assemblage composition to anthropogenic drivers. Methods to characterize assemblages include Bou-Rouch (Figure 4.12a-b; Bou and Rouch, 1967) and vacuum pump sampling (Figure 4.12c; Boulton et al., 1992). Bou-Rouch samples can provide higher abundance and richness estimates, better discriminate between stream types, and achieve greater consistency between replicate samples (Stubbington et al., 2016). This method is thus recommended, although vacuum pump sampling equipment is less expensive and this method can also identify biotic response to abiotic drivers.

Figure 4.12 Equipment and mode of operation for two methods used to pump invertebrates from subsurface sediments: (a) the Bou-Rouch pump and (b) its operation; (c) vacuum-pump sample collection. Further information is provided in Stubbington et al. (2016).

4.5 Semi-aquatic and terrestrial invertebrates

4.5.1 An overview of IRES semiaquatic and terrestrial invertebrate communities

IRES provide a range of flowing, ponded and dry habitats that support diverse communities of aquatic, semiaquatic and terrestrial invertebrates. Communities are typically dominated by ground-dwelling beetles (notably the Carabidae and Staphylinidae; Figure 4.13), ants and spiders, and also include crustaceans, centipedes, millipedes, slugs, springtails, true bugs, true flies and worms (Steward et al., 2017; Stubbington et al., 2019b). Generalists dominate, and specialists with adaptations to tolerate inundation also occur (Larned et al., 2007; Steward et al., 2011). These communities inhabit dry riverbeds, the margins of pools and ponded waters, and – during flowing phases – shorelines, floodplains, exposed gravel bars and unsaturated gravels. Our understanding of IRES terrestrial and semiaquatic invertebrates is limited, with key European studies examining assemblages in France (Corti and Datry, 2016; Corti et al., 2013) and Spain (Sánchez-Montoya et al., 2016).

Figure 4.13 One sampling campaign in two UK karst IRES recorded 23 beetle species in the Carabidae family, including (a) *Bembidion atrocaeruleum*, (b) *Elaphrus riparius*, (c) *B. lampros*, (d) *B. tetracolum*, (e) *Asaphidion curtum* and (f) *B. tibiale*.

Adapted from Stubbington et al. (2018b); © Roy Anderson.

Semiaquatic and terrestrial communities are responsive to changes in both hydrological and geomorphological factors (Steward et al., 2017). However, only one study has assessed their responses to individual stressors: Steward et al. (2018) found that terrestrial invertebrates are responsive to physical substrate disturbance by livestock and feral mammals in dry Australian IRES, and may thus be effective indicators of dry-phase quality. Future research to inform biomonitoring could focus on specific high-potential groups. For example, carabid beetles are common and abundant in aquatic–terrestrial habitats including IRES, and their distribution in relation to abiotic variables may enable managers to distinguish sites of contrasting hydromorphological and physico-chemical quality (Stubbington et al., 2019a).

4.5.2 The MIS-index: an invertebrate index to span the aquatic–terrestrial divide

The MIS-index is a new index developed to describe total invertebrate community response to intermittence (England et al., 2019). It incorporates invertebrate taxa from fully aquatic to terrestrial, all of which are collected during standard biomonitoring surveys done by regulatory agencies (typically 3-minute kick/sweep sampling all wet habitats in proportion to their occurrence). Invertebrate taxa (family, genera and species) are assigned to one of six MIS-groups based on their association with lotic (fast), lotic, generalist, lentic, semi-aquatic, and terrestrial habitats (Figure 4.14). Weighting factors are applied to the richness of each group to give a single score, with different weighting factors used in spring and autumn.

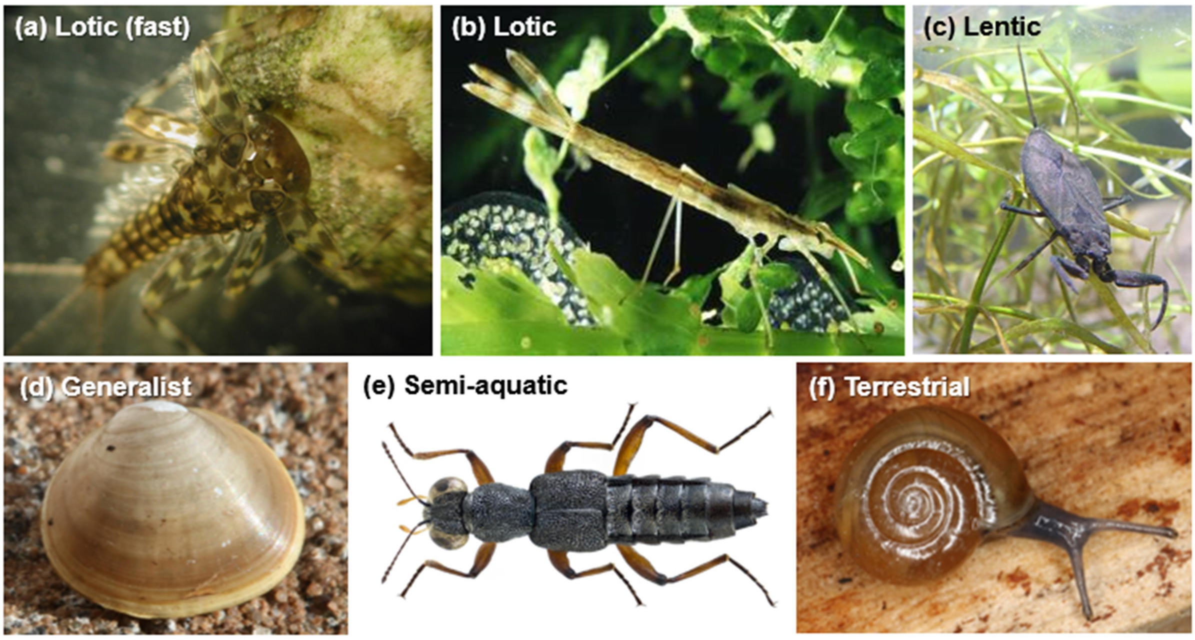

Figure 4.14 A representative species within each of the six MIS-Index habitat association groups: (a) a mayfly characteristic of *lotic (fast)* habitats; (b) a damselfly associated with *lotic* waters; (c) a water scorpion in *lentic* water; (d) a bivalve with *generalist* habitat preferences; (e) a *semi-aquatic* beetle; and (f) a snail indicative of *terrestrial* habitats. For further details, see England et al. (2019).

Preliminary observations indicate that the MIS-index complements existing indices used to assess aquatic invertebrate community responses to drought (i.e. DEHLI; Chadd et al., 2017; section 4.4.3) and to changes in flow (LIFE; Extence et al., 1999), by characterizing responses to flow intermittence and changes in flow state (England et al., 2019). Developed for lowland groundwater-fed streams in south England, the MIS-index requires testing to see how applicable the taxa-habitat associations and weightings are across regions and IRES types. Scientists and managers are encouraged to identify collected semi-aquatic and terrestrial taxa, to provide data to test the index in IRES across Europe. Recognizing and understanding responses to natural intermittence will inform our understanding of IRES biodiversity value and their responses to human pressures that alter ecological quality.

4.6 Aquatic plants

4.6.1 An overview of IRES aquatic plant communities

IRES aquatic plant communities are structured by frequency, magnitude and duration of wet phases, as well as by substrate type, shading by riparian vegetation, sediment depth and water storage capacity, nutrients and oxygen availability (Baldwin and Mitchell, 2000). Due to frequent shifts between flowing, ponded and dry states, IRES communities may not experience succession (Deil, 2005), and only a few strictly aquatic species, such as those with short life cycles or drought-tolerant perennials, form stable, species-poor stands (Rodwell et al., 1995; Sabater et al., 2017). Waterlilies and pondweeds can tolerate only short-term drying, whereas annual duckweeds may occupy mesotrophic to eutrophic slow-moving streams within weeks of flow resumption. Equally, desiccation-tolerant species such as *Ranunculus peltatus* and marginal and wetland plants can dominate summer-dry reaches in 'winterbourne' IRES (Westwood et al., 2006a,b).

Figure 4.15 Changes in vegetation at a single site on a 'winterbourne' IRES in south England: (a) aquatic macrophytes dominate during high flows in early spring; (b) marginal then semi-aquatic species encroach during flow recession pre-drying in early summer; (c) terrestrial plants dominate during the dry phase in late summer and (d) persist as water levels increase in autumn.

Aquatic plants should be surveyed seasonally in different phases of the growing period and should follow wet-dry habitat cycles (Figure 4.15; Holmes et al., 1999; Westwood et al., 2006a). Sampling of vegetation plots or transects in 100-m IRES reaches together with assessment of hydromorphological habitat characteristics (Raven et al., 1997) can be an appropriate survey method for ecological assessment (Hughes et al., 2010). Macrophyte trophic indicator values calibrated for rivers in specific regions (such as the Mean Trophic Rank in UK; Dawson et al., 1999), as well as general metrics, such as species richness, diversity indices and functional macrophyte groups can be suitable response variables that reflect both natural habitat complexity and human impacts (Kail et al., 2015).

4.6.2 Performance of a standard aquatic plant-based index in UK chalk streams

Mean Trophic Rank (MTR; Holmes et al., 1999) is a standard UK biomonitoring method that uses percentage cover of aquatic plants (typically identified to species) to assess nutrient status. However, because MTR is based solely on in-channel aquatic plants, it is of limited use in dry IRES colonized by terrestrial species. A modified version of MTR – with additional recording of non-aquatic herbs and grasses – was thus used to assess the responses of plant assemblages in groundwater-fed IRES in south England to abiotic variables indicative of specific human impacts: bank slope, livestock poaching, sediment heterogeneity, shade and water quality (Stubbington et al., 2019a). Distinct assemblages were associated with different impacts, with higher species richness, diversity and plant cover indicative of less impacted sites (Figure 4.16). However, the modified MTR did not distinguish between sites with *good* and *poor* water quality, which may be because water was assessed during wet phases. Overall, plants were judged to be high-potential indicators of dry-phase biological quality. However, terrestrial plants were identified to a very coarse level, and community responses could be characterized more effectively if these taxa were also identified to species level. Research is needed to devise a method to assess biological quality based upon the whole plant assemblage, encompassing species from fully aquatic to fully terrestrial.

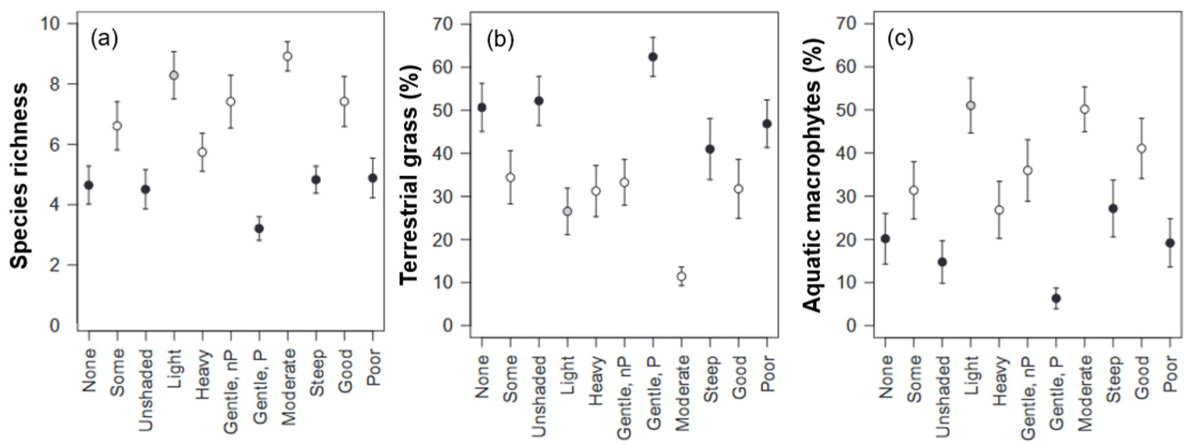

Figure 4.16 Mean ± 1 SE metrics for plant assemblages surveyed in the dry channels of six rivers in relation to five aspects of ecological quality: sediment heterogeneity (none, some), the extent of shading (unshaded, light, heavy), bank slope (gentle, moderate, steep) and poaching (nP, not poached; P, poached), and water quality (good, poor): (a) species richness; (b) terrestrial grass cover (%); and (c) aquatic macrophyte cover (%). Black, grey and white-filled symbols indicate impacted, semi-impacted and unimpacted conditions, respectively.

Adapted from Stubbington et al. (2019a) http://creativecommons.org/licenses/by/4.0

4.6.3 A new plant-based aquatic–terrestrial index designed for IRES

A new index, the Plant Flow Index (PFI; Westwood et al., in prep.), has been developed for use in groundwater-fed lowland streams in south England. The PFI currently includes 46 species and taxa covering a range from obligate aquatic to fully terrestrial plants, with another 80 species with lower average abundances under consideration. Species are abundance-weighted and coded according to their tolerance of channel drying, as determined by statistical responses to hydrological metrics including no-flow metrics. An aggregation of the scores gives a single value for a site, with lower scores indicative of greater flow intermittence and higher scores of more perennial conditions. Like the invertebrate-based BIODROUGHT (see section 4.4.3) and DEHLI indices (see section 4.4.4), the PFI can thus inform interpretation of biomonitoring data collected to assess IRES biological quality.

The PFI was developed and tested on groundwater-fed streams and responds to intermittent and ephemeral flow regimes. PFI responses to hydrological metrics calculated for differing time spans were further compared to other plant-based indices, and the index found to be more statistically robust than any other index. The current species list can be updated and adapted to suit other geologies and locations, and the PFI is therefore a flexible system with a potential for application beyond UK lowland IRES.

4.7 Semi-aquatic and terrestrial plants

4.7.1 An overview of IRES semi-aquatic and terrestrial plant communities

IRES support semi-aquatic and terrestrial plants including mosses, herbs and grasses. These plants occur instream during dry phases, in marginal areas, on the banks, and extend into the riparian zone (Bruno et al., 2014; Sabater et al., 2017). All IRES plants experience wet-dry cycles, and terrestrial species gradually replace aquatic species during dry phases (Figure 4-15, Figure 4-17). Riparian vegetation in IRES is characterized by short-lived shrubs and trees with shorter canopies and longer roots, whereas tall, fast-growing, large-leaved trees and shrubs, and perennial herbaceous species typically line perennial rivers (Bagstad et al., 2005; Dodds et al., 2004; Stromberg and Merritt, 2016). Canopies thus become sparser as intermittence increases, from riparian gallery forests in perennial streams to more open riparian woodland adjacent to ephemeral streams. Riparian species richness (i.e. alpha diversity) may decline with intermittence, but variation among sites (i.e. beta diversity) may peak at IRES sites due to variability in moisture conditions (Olson et al., 2001; Katz et al., 2012). We know very little about the semi-aquatic and terrestrial plant communities that colonize channels during dry phases, with preliminary observations indicating that diverse, grass-dominated assemblages soon establish (Figure 4.17; Stubbington et al., 2018b).

Figure 4.17 The semi-aquatic and terrestrial plant communities in dry 'winterbourne' chalk IRES sites in south England (a-c) are dominated by different mixes of grasses, rushes and broad-leaved ruderals.

IRES instream and riparian plant communities may be more affected by human activities that alter flow regimes compared to those of perennial rivers (Bruno et al., 2016b). IRES riparian species are adapted to natural flow variability, but human-induced increases in drying may act as a disturbance to which species are not adapted (Stanley et al., 2004). For example, increasing intermittence in the Mediterranean Basin has caused plant communities to become more terrestrial (Gudmundsson et al., 2019). In contrast, dam management, wastewater effluents, inter-basin transfers, and agricultural, urban and industrial runoff can increase summer flow, transforming IRES into perennial streams (Hassan and Egozi, 2001; Sabater et al., 2017) and thus decreasing habitat availability for terrestrial and semi-aquatic plants (Datry et al., 2014). Impacts of such ‘perennialization’ on plant communities have yet to be characterized.

Stubbington et al. (2019a) showed that instream dry-phase plant communities respond to non-hydrological human impacts, namely sediment composition, shading, poaching and geomorphological impacts (as bank slope). Identifying non-aquatic plants only as “grasses” or “herbs”, this study (using data from Holmes [1999] and Westwood et al. [2006a]) indicated these groups' potential as bioindicators. Species-level characterization of the terrestrial and semi-aquatic plant assemblages that establish in dry channels is needed to inform their use in dry-phase biomonitoring.

4.7.2 Riparian plant communities as biomonitors of Mediterranean IRES: a case study

Bruno et al. (2016a) investigated the use of riparian vegetation as bioindicators in Mediterranean-climate perennial rivers and IRES using taxonomic and functional community metrics. Riparian species richness and quality (Riparian Quality Index, González del Tánago and García de Jalón, 2011) were reduced by land-use intensification, flow regulation and natural flow intermittence. However, functional redundancy (which represents the number of species that make similar contributions to an ecosystem function) responded to multiple abiotic stressors (Bruno et al., 2016b) and discriminated between different categories of human impact intensity in perennial reaches and IRES. This allowed setting of thresholds to identify, predict and map impact levels in both stream types (Figure 4.18; Bruno et al., 2016a).

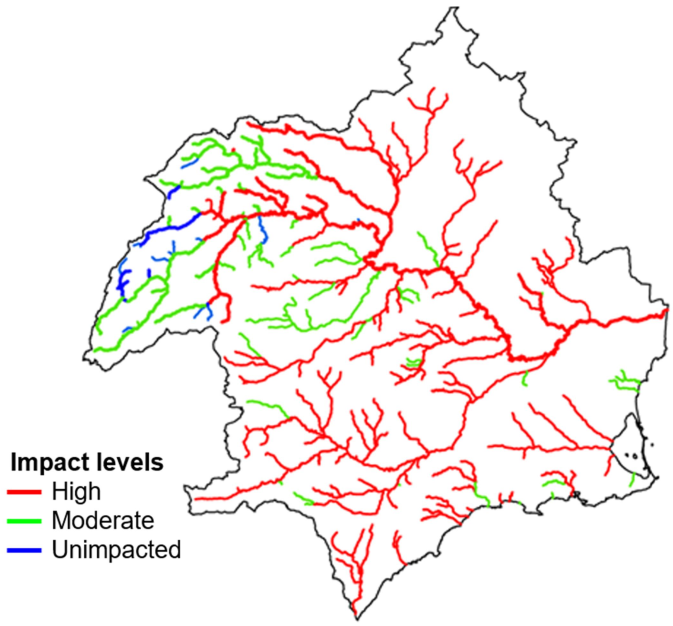

Figure 4.18 Spatial patterns of human impact levels based on predicted values of the functional redundancy of riparian vegetation communities in IRES (thin lines) and perennial rivers (thick lines) in a basin in south-east Spain. Bruno et al. (2016a) provide further details.

From Bruno et al. (2016a).

Community-based functional metrics may be more informative than taxonomic metrics because they enable greater inter-taxon and inter-region comparability (McGill et al., 2006). Functional approaches including redundancy may thus complement taxonomic metrics by assessing ecosystem function as well as discriminating overall and specific human impact levels. These metrics are simple and low-cost to implement, requiring no additional fieldwork or expertise beyond that used to obtain taxonomic information. However, in all cases, biomonitoring of IRES semi-aquatic and terrestrial riparian and instream vegetation requires sufficient spatial and temporal replication to describe the variability in their flow regimes.

4.8 Microorganisms

4.8.1 An overview of IRES microbial communities

Freshwater microbial communities include archaea, bacteria, protozoans, fungi, cyanobacteria and algae. These microbes perform processes such as nutrient cycling and thus support ecosystem functioning. In freshwaters including IRES, microbes cover benthic and hyporheic substrates, creating biofilms that act as metabolic 'hot spots'. Diatoms are particularly diverse, abundant, and thus well-studied contributors to biofilms (Figure 4.19; Makovinska and Hlubikova, 2014). These assemblages differ between IRES and perennial streams, with lower species richness typical in IRES (Tornés and Ruhí, 2013). While IRES support desiccation-tolerant species, drying-sensitive species are absent and may be threatened by an increase in intermittence (Falasco et al., 2016a). Assemblages also vary between wet and dry phases in IRES, due to changes in water availability and physico-chemistry (Sabater et al., 2017).

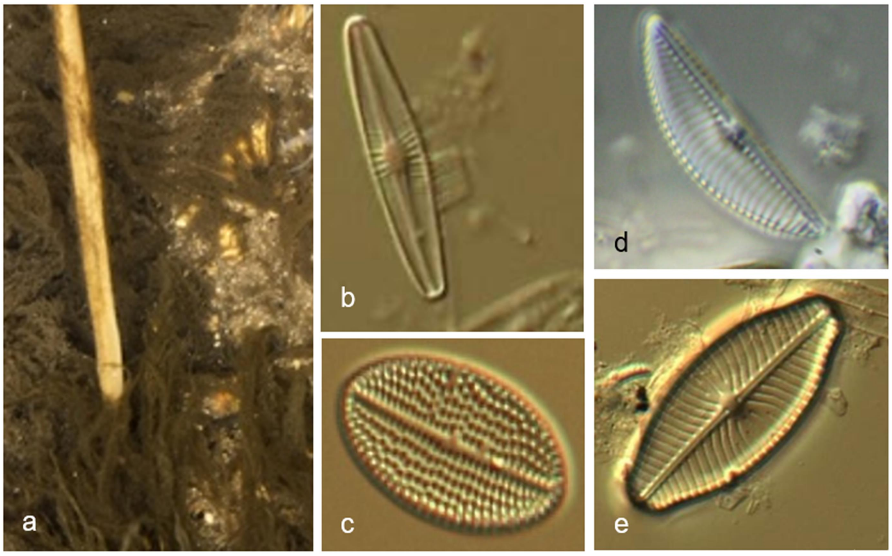

Figure 4.19 The biofilm coating IRES surfaces include a diverse diatom community, including the species (b) *Achnanthes minutissima*, (c) *Amphora fogediana*, (d) *Cocconeis placentula* and (e) *Placoneis gastrum*.

© Judy England (a) and P.J. Meadows (b-e).

During wet phases, diatoms can act as bioindicators of water quality, due to their rapid, species-specific responses to abiotic variables including nutrients. Accordingly, diatom indices including the Specific Polluosensitivity Index (IPS; Coste, 1986) and the Trophic

Diatom Index (TDI; Kelly and Whitton, 1995) have been developed to characterize responses and thus assess trophic status in perennial streams. However, diatom communities also vary in response to climatic and other abiotic factors (Pajunen et al., 2016; Soininen et al., 2019) and thus index performance may vary both among regions, and between IRES and the perennial streams in which they were developed. For example, the IPS has been evaluated as suitable for IRES in Cyprus (Montesantou et al., 2008) but not Croatia, where the TDI *is* effective (Mihaljević et al., 2011; see Table 2, Stubbington et al., 2018a). Where current approaches are ineffective in IRES, functional metrics as well as IRES-specific taxonomic indices warrant exploration (Falasco et al., 2016b).

4.8.2 Performance of a standard diatom index in temperate IRES

The ecological quality of IRES is typically assessed during flowing phases using indices developed for perennial systems. The ability of such indices to characterize IRES quality varies during wet phases, and index performance during dry phases remains all but unknown. To address this research gap, Stubbington et al. (2019a) evaluated the indicator potential of dry-phase diatom assemblages from sites with high, good and moderate WFD ecological status, with deviations from high status reflecting enrichment by inorganic nutrient. Biofilm samples were collected on 1-3 dates during single, continuous dry phases from six sites across five rivers in the Adour-Garonne catchment, France. The sites experienced dry phases of 4-30 weeks. Sampling methods were based on French national standard (AFNOR, 2014a,b), with some adaptations to suit dry-phase conditions.

Diatom assemblages from high-status sites differed from those from more impacted sites and were more heterogeneous (Figure 4-20a). Comparison with samples collected during preceding and subsequent flowing phases indicated that 81% of the assemblage remained viable during drying events, although persistence depended on the dry-phase duration. Among drying-tolerant taxa, high abundance of the *Achnanthidium minutissimum* complex was a significant indicator of high status (Figure 4-20b), and 16 other taxa were restricted to high-status sites. Equally, six taxa were indicative of good-to-moderate status, including *Amphora pediculus*. French Biological Diatom Index (AFNOR, 2007) scores were higher at unimpacted sites than at nutrient-enriched sites, and reached values comparable to those at perennial sites of equivalent status. Diatoms may therefore have potential to indicate IRES quality during dry phases, but the preliminary case study presented by Stubbington et al. (2019a) was based on only a small data set. Further research is thus needed to better characterize temporal change in assemblage composition during dry and wet phases, to inform the potential future use of diatom assemblages as dry-phase indicators.

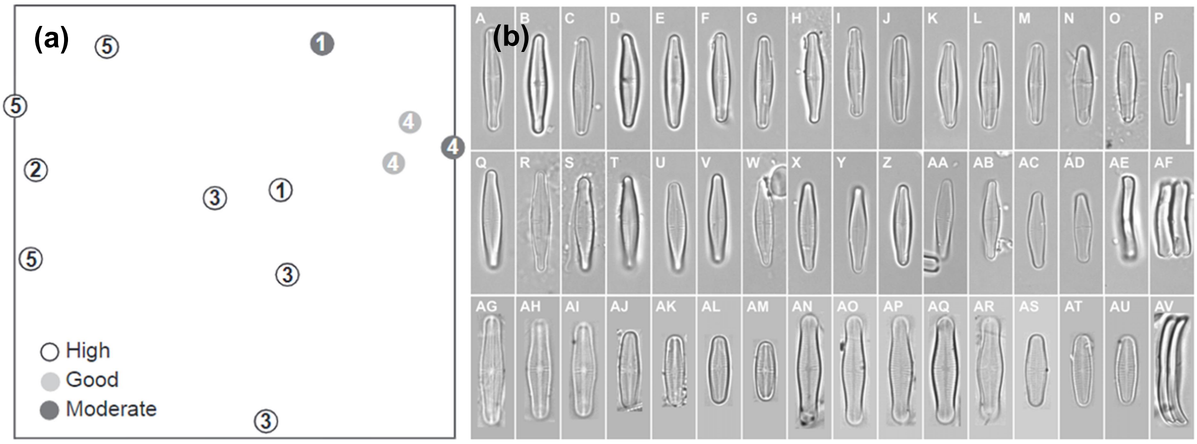

Figure 4.20 (a) Multivariate ordination indicating differences in the composition of diatom communities from dry sediment samples from *high*, *good* and *moderate* WFD ecological status sites across five IRES (1–5); (b) morphological diversity in the *Achnanthidium minutissimum* complex, identified as indicative of high-status sites by Stubbington et al. (2019a).

(a) Adapted from Stubbington et al. (2019a); (b) © LM images: Carlos E. Wetzel.

4.9 Future directions: beyond the taxonomy of local communities

4.9.1 Introduction

IRES biomonitoring is done by morphological identification of the species within assemblages sampled at local (within-reach) spatial scales. Across and beyond Europe, such an approach is also typical in perennial systems, and undoubtedly enables accurate ecological quality assessments (Birk et al., 2012). However, as researchers and managers collaborate to improve their biomonitoring and management, we highlight innovative approaches that could enable robust characterization of IRES quality. Genetic tools are gaining prominence across ecosystems and have particular potential in IRES due to their potential concurrent characterization of aquatic and terrestrial biodiversity contributions. Functional (trait-based) approaches are also the subject of increasing interest, and have potential for widespread application due to their independence from regional species pools. Regardless of whether the approach is genetic, structural (i.e. taxonomic) and/or functional, considering local communities within a wider, metacommunity context is particularly important in IRES, due to the importance of dispersal processes in enabling recolonization by aquatic and terrestrial biotas after wet and dry phases commence, respectively.

4.9.2 Genetic tools to assess biodiversity across aquatic–terrestrial boundaries

Genetic tools are emerging as a promising way to assess biodiversity. Species composition can be inferred by metabarcoding either organismal DNA in bulk samples (e.g. see Hajibabaei et al. [2011] for macroinvertebrates, Kermarrec et al. [2013] for diatoms) or eDNA in water or sediment samples (e.g. Macher et al., 2018). Both methods use High Throughput Sequencing (HTS) to concurrently generate millions of DNA sequence reads belonging to hundreds of samples. Metabarcoding targets a DNA ‘barcode’: a specific genome region, which varies depending on the biotic group (Hebert et al., 2003). Species are identified by matching the retrieved unknown sequences with known sequences linked to a taxonomy in reference databases.

Compared to morphological identification, metabarcoding has advantages including better reproducibility and greater replication, due to rapid sample processing. Another advantage is the finer-resolution identification, not least for groups with complex taxonomy, juvenile stages without diagnostic features and damaged specimens (Elbrecht et al., 2017). Genetic methods also improve biodiversity estimates by identifying cryptic species, which are hard to detect morphologically (Kahlert et al., 2019). Metabarcoding also has drawbacks. It cannot reliably estimate abundance for groups including macroinvertebrates (e.g. Elbrecht and Leese 2015), although presence-absence data can enable biomonitoring of such groups (Beentjes et al., 2018), and abundance can be estimated for groups including diatoms (Vasselon et al., 2018). Second, gaps in reference databases limit species-level identification, especially for groups not targeted by barcoding projects; availability of complete databases is required for taxonomic assignments that inform reliable ecological assessments (Weigand et al., 2019). Lastly, laboratory biases occur, but should decline as methods are standardized (Leese et al., 2018).

The finer taxonomic resolution achieved with genetic techniques will improve our ability to identify natural/anthropogenic impacts on biotic communities, because more precise species-specific sensitivity scores can be obtained for different stressors. For instance, cryptic species of the mayfly genus *Deleatidium* identified with genetic techniques showed contrasting responses to nutrient and sediments levels in New Zealand rivers (Macher et al., 2016). Furthermore, haplotypes (a measure of the relatedness among individuals of the same species, which can be inferred from metabarcoding data; Elbrecht et al., 2018) could be used to characterize dispersal patterns of organisms and to detect genetic ‘bottlenecks’ (in which population size and thus genetic diversity decline sharply due an event such as a drought) in IRES, thus informing management actions taken to conserve target species.

In IRES, metabarcoding could enable biomonitoring of terrestrial and aquatic biodiversity during wet and dry phases. During dry phases, DNA in sediments, biofilms, eggs or other biotic substrates can be extracted and analysed, targeting either specific groups (e.g. diatoms) or a wider diversity of taxa (e.g. metazoan families and orders). Based on metabarcoding data, it should be possible to identify organisms that can survive dry phases and also to design new indices to monitor the ecological status of IRES (Pawlowski et al., 2018; Hering et al., 2018).

4.9.3 Can a trait-based approach indicate IRES ecological quality?

Functional approaches – which explore species traits, not their names – represent a useful approach to study the effects of multiple stressors on river ecosystems (Dolédec et al., 1999). However, assessment of functional community responses to combined effects of natural disturbances and human impacts remain limited. To date, using functional approaches, Belmar et al. (2019) distinguished biotic responses to flow intermittence and flow regulation; Gutiérrez-Cánovas et al. (2019) detected human impacts along a natural salinity gradient; and Bruno et al. (2016) identified metrics that differentiate riparian plant community responses to natural intermittence and human impacts (see section 4.7.2). However, Soria et al. (2020) are the first to use functional metrics to distinguish responses to drying and to multiple human impacts with IRES.

Soria et al. (2020) sampled aquatic macroinvertebrates in Mediterranean IRES along gradients of natural flow intermittence and human impacts, to examine their combined effects on traditional (taxonomic) biotic indices and novel functional metrics, including functional redundancy (FR) and response diversity (RD). Here, FR describes the number of species in a functional group, and RD indicates the extent to which functionally similar species vary in their response to environmental changes (Figure 4.21; Suding et al., 2008; Laliberté et al., 2010). Only one taxonomy-based index responded to both intermittence and human impacts, but several functional metrics detected impacts regardless of intermittence, and the community FR was even effective for assemblages sampled from isolated pools.

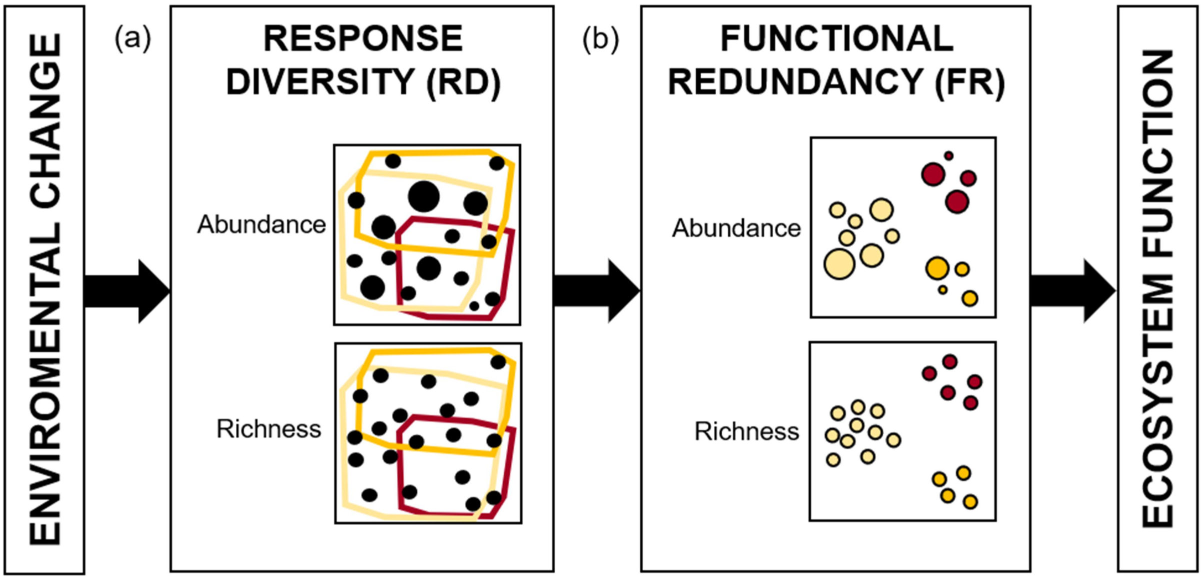

Figure 4.21 A conceptual diagram illustrating two functional metrics: (a) response diversity (RD) and (b) functional redundancy (FR), with colours indicating different functional groups. In (a), polygons surround the mean abundance (top) or richness (bottom) of three functional groups with traits that respond differently to environmental change, i.e. polygon size is proportional to RD. In (b), species abundance (top) and richness (bottom) differs among three functional groups with traits that have different effects on ecosystem function, i.e. the number of symbols per group is proportional to FR.

Further research is needed to identify functional summary metrics that describe biotic responses to specific human stressors in IRES. Choosing suitable traits is vital, because it can determine the reliability of calculated summary metrics such as functional redundancy and diversity, with emerging evidence suggesting descriptors of body size, resistance forms and diet as informative (Wilkes et al., in prep). However, large-scale selection for certain traits by environmental drivers (e.g. desiccation-resistant forms in drier climates) may result in regional differences in informative traits.

4.9.4 Using a metacommunity perspective to enhance IRES biomonitoring

A metacommunity comprises a set of local communities connected by dispersal (Leibold et al., 2004). Metacommunity theories assume that assembly of local communities is regulated by environmental filtering (i.e. species occur in habitats that match their environmental preferences) and dispersal (i.e. species occur in habitats they can reach, which depends on connectivity and species-specific dispersal ability). IRES support dynamic metacommunities in which local communities experience alternating wet and dry phases as well as changes in hydrological connectivity, creating mosaics of habitats that repeatedly connect and disconnect (Datry et al., 2016). In such mosaics, recolonization after changes in flow state is driven by dispersal from source communities; for example, after flowing phases resume, aquatic organisms recolonize IRES from perennial reaches (Bogan and Boersma, 2012). Recognizing dispersal and connectivity thus enables prediction and understanding of the species present in a local IRES community at any given time (Sarremejane et al., 2017).

Biomonitoring data are typically interpreted assuming that local abiotic conditions explains community composition, whereas explicitly consideration of the effects of dispersal and connectivity is rare (Heino, 2013). This may explain why biomonitoring methods sometimes perform poorly in dynamic ecosystems such as IRES. For example, if dispersal is very high, species can be found in degraded sites, and if dispersal is low, species may not occur at sites with abiotic conditions that match their preferences. To improve understanding of biomonitoring data, connectivity could be measured as potential dispersal routes (e.g. distance to the nearest perennial site) and barriers to of recolonist sources, and dispersal capacity could be examined by allocating species to groups based on dispersal strength (Heino et al., 2017). New methods that integrate metacommunity perspectives into IRES management are in development, for example a dispersal trait database (Sarremejane et al., in prep.) will enable evaluation of the effects of invertebrate species' dispersal capacity on metacommunity organization. Second, Cid et al. (in prep.) are developing a broad conceptual basis of how to include dispersal and connectivity in IRES biomonitoring.

4.10 Take-home messages

IRES represent a high and increasing proportion of river networks in many European regions. Water managers require approaches to enable efficient assessment of their ecological status, and thus meet the requirements of national and international legislation, notably the WFD (see section 4.2). This chapter highlights best practice from across Europe, including notable progress in the Mediterranean regions in which IRES dominate networks, as well as in cooler, wetter temperate regions. Despite these advances, environmental managers continue to face considerable challenges in conducting effective ecological status assessments in IRES, both across and beyond Europe (see section 4.2.3). Below, we provide a set of take-home messages to inform effective biomonitoring and major knowledge gaps that represent priorities for future research and development.

- Testing and evaluation of indices developed to assess the ecological quality of perennial streams is required in the IRES of many countries (Table 4.1). This testing needs to recognize variability among IRES and their ecological communities, both among countries and among national subtypes; index performance is likely to vary among IRES.
- Only where the performance of a standard index has been shown to be effective in IRES should it be used to assess ecological quality. In such cases, temporal variability in the community composition must be recognized and surveys timed to represent peak biodiversity, e.g. after a long enough flowing phase for aquatic communities to have recolonized. Index performance is likely to vary both within and between years.
- Where perennial indices are shown to be inadequate, managers should first confirm that communities were sampled at an appropriate time, e.g. after a long enough flowing phase. If so, new indices may require development, which should be informed by characterization of the communities indicative of unimpacted conditions in a particular IRES subtype. However, spatial and temporal variability in community composition can impede description of reference conditions in IRES.
- Indices developed to characterize community responses to changing instream conditions (including ponding and drying; see sections 4.4.3, 4.5.2) can support interpretation of data collected during ecological quality assessments. Such indices

have the potential for widespread use beyond their country of origin, but require testing and adaptation to reflect the species and habitats present in different regions.

- The biotic groups used as biomonitors in perennial streams vary in their bioindication potential in IRES. For example, fish communities in many unimpacted IRES are too species-poor to enable inference of ecological quality, whereas diatom communities often remain effective as biomonitors or trophic status, if sampled during flowing phases.
- Where there are no suitable approaches based on aquatic biota, for example in ephemeral streams with short, unpredictable flowing phases, the development of new approaches is a priority. Multiple terrestrial communities, including invertebrates (see section 4.5.1) and plants (see Section 4.6.3), show high potential. In the meantime, assessments based on hydromorphology can indicate the naturalness of channel processes (see Chapter 2), thus enabling inference of ecological quality.
- Emerging tools that may enable assessment of IRES biodiversity across aquatic–terrestrial boundaries include genetic profiling (see section 4.9.2). Functional characterizations based on species' traits (see Section 4.9.3) and consideration of metacommunity dynamics (see Section 4.9.4) are two other innovative approaches that may inform understanding of IRES ecological quality. Managers need to contribute to the testing and evaluation of these approaches, to facilitate their application to ecosystem management.

Table 4.1 Performance in IRES of biotic indices developed to assess ecological quality in perennial rivers. Performance evaluated in IRES with long, seasonal flowing phases. Replacement indices (for those evaluated as not suitable) have been tested by or are in use by Water Framework Directive competent authorities. Adapted from Stubbington et al. 2018a.

WFD GIG	Country	Index	Biotic group	Suitable?	Replacement	Source
Eastern Continental	Croatia	IBMWP, SI_{HR}	Macro-invertebrates	Yes		See Stubbington et al. (2018a)
		IPS	Diatoms	No	TDI	
Mediterranean	Cyprus	STAR_ICMi	Macro-invertebrates	Yes		Buffagni et al. (2012)
		IPS	Diatoms	Yes		Montesantou et al. (2008)
		IBMR	Macrophytes	No	MMI	Papastergiadou and Manolaki (2012)
	Greece	HESY-2	Macro-invertebrates	Yes		Lazaridou et al. (2016)
	Portugal	IPtIS, IPtIN	Macro-invertebrates	Yes		INAG (2009)
		IPS	Diatoms	Yes		EC (2012), in Skoulikidis et al. (2017)
	Spain – Catalan RBD	IBMWP	Macro-invertebrates	No	IMMi-T; IMMi-L	Munné and Prat (2011)
		IPS	Diatoms	Yes		See Stubbington et al. (2018a)
North East Atlantic	UK	BMWP, ASPT	Macro-invertebrates	No	Required	Wilding et al. (2018)
Northern; Central / Baltic	France	I_2M_2	Macro-invertebrates	No	Required	Pelte et al. (2012, 2014)

Abbreviations: BMWP, Biological Monitoring Working Party; HESY-2, Hellenic Evaluation System; I_2M_2, French macroinvertebrate multimetric index; IBMWP, Iberian BMWP; IBMR, L'Indice Biologique Macrophytique en Rivière; IMMi-L (Iberian Mediterranean Multimetric Index [IMMi] – qualitative); IMMi-T (IMMi – quantitative); IPS, Indice de Polluosensibilité Spécifique; IPtIN (Invertebrate Index for northern Portugal); MMI, Multimetric Macrophyte Index; IPtIS (Invertebrate Index for southern Portugal); TDI, Trophic Diatom Index; SI_{HR}, Croatian index; RBD, River Basin District; STAR_ICMi, STAndardisation of River classifications Intercalibration Common Metric index; WFD GIG, Water Framework Directive Geographical Intercalibration Group.

5. Ecosystem services and social perception

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5.1. Introduction

5.1.1 In a nutshell

- There is a variety of benefits that IRES provide to our societies, from the provision of materials such as water and timber, to iconic species, the regulation of biogeochemical cycles, and space for cultural manifestation and as a corridor for both wild and herded animals.
- Drying and rewetting processes, timing and duration of different aquatic phases, have an effect on the biodiversity and ecosystem functioning, as well as on the provision of ecosystem services and on the social perception of them.
- There are intrinsic and relational values associated to IRES that are not usually recognised, including sense of place, cultural identity, social cohesion or nature stewardship.
- There is a long list of indicators that can be used to assess the provision of ecosystem services, and different techniques of monetary and non-monetary methods can be applied to assess their value.
- Public participation is also necessary to understand the multiple values of IRES and to improve social perception. Participatory mapping, citizen science, and scenario planning are some of the methodologies can be employed.

5.1.2 The importance of accounting the value of ecosystem services of IRES

The most complete definition of ecosystem services is “the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life” (Daily G. C, 1997). But they can also be defined as simple as “the benefits people obtain from ecosystems” (M.A., 2003). Building on these definitions, ecosystem services research has spread and increased across different research fields and disciplines and incorporated multiple methodologies and approaches; in the case of river ecosystems they have been tentatively used in river conservation and restoration practices (Martin-Ortega et al., 2015). In fact, environmental managers, when focusing on the provision of ecosystem services – in addition to biodiversity and ecosystem function –, highlight the importance of maintaining and improving human livelihoods and well-being. Ecosystem services-based approaches promote holistic management that allows the coexistence of multiple ways of using and enjoying a river with good ecological status. By using such approaches, managers not only improve the living conditions of people, but they also promote social acceptance of the environmental policy and management, and thus reduce social tensions and conflicts. At EU level, the importance of considering ecosystem services is highlighted by the Biodiversity Strategy which called on Member States to map and assess the state of ecosystems and their services in their national territory. They must also assess the economic value of such services and promote the integration of these values into accounting and reporting systems at EU and national level by 2020.

In this chapter, we focus on the ecosystem services provided by IRES, their biophysical conditions and people's perceptions and values, as well as methods for assessing them from supply and demand points of view.

5.2 Ecosystem services of IRES

Ecosystem services depend, not only on the ecosystem that provides them, but also on the society that values and benefits from them. Ecosystem services are by definition context-dependent. This means that the ecosystem services provided by Mediterranean, Alpine and Continental IRES will differ not only because of differences in their biophysical attributes, but also because of dissimilar and fluid socio-cultural contexts. This section provides an overview of ecosystem services that are common to all types of IRES. However, any assessment that aims at using an ecosystem services-based approach needs to first identify the main actors and stakeholders: "who benefits?", "are there any losers?", and "what services do they perceive and value?". Of course, the local people may benefit or lose from IRES management, but some ecosystem services benefit all humans on Earth (e.g., carbon sequestration that reduces greenhouse gases). This can be done by doing preliminary literature research, by observation, by asking a group of experts, key stakeholders, and/or by developing a survey that aims to reveal people's perceptions and values.

In the following subsections, a description of the ecosystem services in IRES is presented according to their usual classification into **Provisioning**, **Regulating** and **Cultural** ecosystem services. The Common International Classification of Ecosystem Services (CICES) has developed a more exhaustive classification of services that we are not following here, but which is available at *https://cices.eu*. For an in-depth description of the services provided by IRES, you can also consult Stubbington et al. (resubmitted).

5.2.1 Provisioning ecosystem services

Provisioning services are the products directly obtained from ecosystems. They usually have a direct and consumptive use, which means that the enjoyment of those services usually requires the consumption of a good. They are also the easiest type of ecosystem services to assess in monetary terms, since those goods are sometimes already marketed. Consequently, these ecosystem services are relatively well understood and recognised and have usually been promoted at the expense of river health and other types of services.

The **provision of freshwater** by IRES is crucial for supplying drinking water and for maintaining agriculture, farming and industry, especially in arid and semi-arid zones, where permanent water courses are scarce or even absent. This service is consistent with the flow regime and the aquatic states, being the eurheic and hyperrheic the most recommendable for water abstraction (see chapter 2 for more information). Moreover, some IRES are connected to aquifers, being able to provide freshwater if groundwater is present. To increase the provision of water when the needs are the highest, during the dry season, some management practices include the artificial recharge of aquifers, the storage in off-channel reservoirs (see Figure 5.1), and the use of efficient techniques of water use (e.g. drip irrigation).

IRES can also provide food in diverse ways. Fishing takes place during wet phases, although aestivating fish can also be captured by excavation during dry phases in arid regions. Hunting is a more common activity, since IRES are habitat for waterfowls. As providers of space for rearing animals, IRES are used as corridors for livestock shepherds, but it's rare to have farming facilities installed in the river floodplain due to the unpredictability of flow regime. **Food provision** also includes the cultivation of crops (in the floodplains, but also in the river channel) and the collection of wild plants (e.g., blackberries, in Figure 5.2).

Figure 5.1 Freshwater provision. Water irrigation pond in Iruraitz-Guana, Spain, that is fed by an IRES.

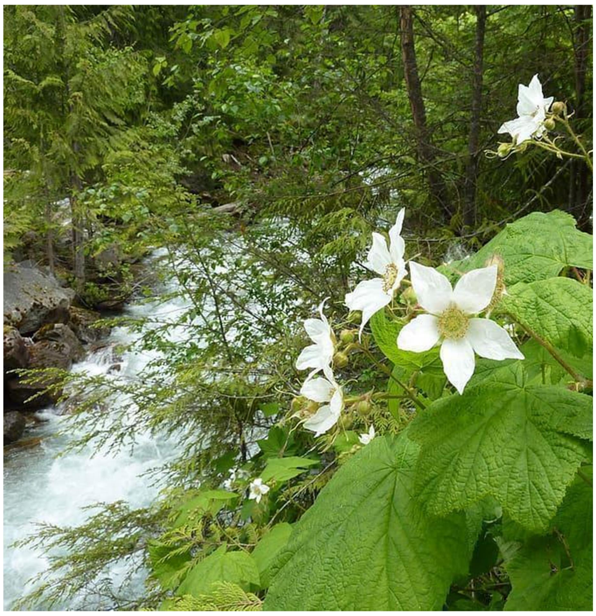

Figure 5.2 Food provision. Blackberries growing close by a stream.

Figure 5.3 Provision of raw materials. In the picture we can see tons of gravel accumulated in the riverbanks of an IRES (©Iakovos Tziortzis).

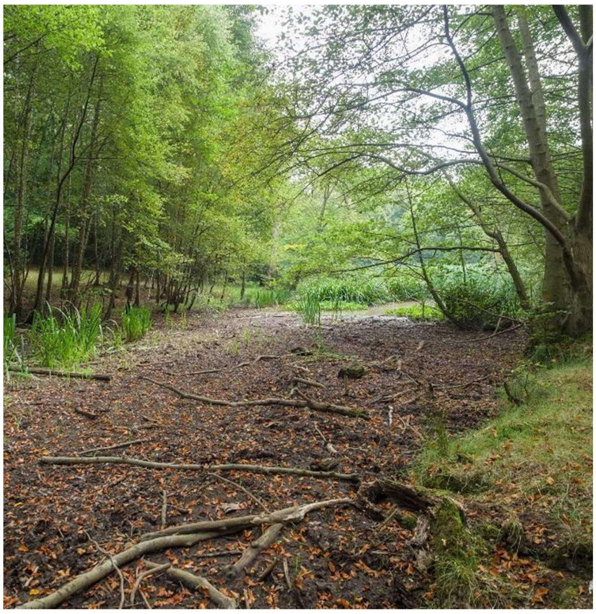

Figure 5.4 Climate regulation. Organic matter is accumulated during the dry periods of this creek in the Burnham Beeches, United Kingdom, having an effect on carbon sequestration and climate regulation.

The **provision of raw materials** is based on the extraction of plants and inert materials i.e. gravel and sand for construction (see Figure 5.3), timber, fuelwood, decoration, and other multiple purposes. As well as perennial rivers, IRES provide habitat for species that are very often employed in different socio-cultural contexts (e.g., *Salix*, *Populus*), and also sediment for different purposes. In any case, intermittence does not seem to have a direct significant effect on the provision of those materials.

IRES play a critical role in the **provision of genetic resources**. It is generally believed that pools promote genetic diversity, resistance and resilience of, e.g., invertebrates and fish populations or even semi-aquatic vertebrates; and dry riverbeds promote communities that use the dry channels either as dispersal corridors or as seedbanks. Moreover, organisms inhabiting IRES experience extreme changes in flow conditions, from drying during summer to flooding during winters which lead to new adaptations, e.g. dispersal forms and resistance strategies that would promote recolonization. Those adaptations of, for instance, desiccation-tolerant invertebrates and plants that colonize during dry phases, are based on molecular strategies to protect against dehydration that may offer new opportunities of IRES for the **provision of biochemical products** that benefit human wellbeing (Stubbington et al., submitted).

5.2.2 Regulating ecosystem services

Regulating services are those benefits provided by ecosystem processes that moderate natural phenomena. They usually have an indirect use value, and it is common that people do not perceive those intangible benefits they receive from regulating services because they are difficult to recognize.

Climate regulation explains the capacity of the ecosystem to buffer local climate conditions. In arid or semiarid zones, where trees are scarce, they are concentrated in IRES hence providing shed and causing a cooling effect. This benefits animals (e.g., livestock) and humans. Climate regulation also means buffering climate change effects, and it is especially linked to carbon sequestration. However, the carbon budget of IRES is not well understood. On the one hand, the capacity of streams to retain organic matter is enhanced during the dry phases when carbon accumulates and decomposition slows down (von Schiller et al. 2017; see Figure 5.4), On the other hand, dry riverbeds can emit large quantities of carbon dioxide (Marcé et al. 2019). These processes may be altered by the presence of dams and woody debris in the river channel.

The **regulation of air quality** consists of the retention of pollutants by plants and microbes of the ecosystem; the improvement of the air quality brings pervasive effects on human health. In this sense, the presence of riparian forest will improve air quality by intercepting air pollution and absorbing gaseous pollutants through leaf stomata. This is however a general service provided by both perennial rivers and IRES.

The **regulation of nutrient cycling** – including water purification – by streams and rivers relies on transport (by the water flow) and residence time of water and solutes (defined by geomorphology), and biological and chemical retention of nutrients. Ecosystem services are linked to two different, interrelated aspects: on the one hand, to the continuity and balance of the global nutrient cycles; and, on the other hand, to water security (provisioning and quality) by reducing eutrophication. Different elements of IRES that favour nutrient cycling are the drying-wetting oscillations, the diversity of geomorphological elements, and the presence of aquatic plants, biofilms, and riparian forest.

Regulation of water flow and protection against extreme events is very important in IRES due to their variability and unpredictability inherent. The variability of the hydrological phases allows storing water within the floodplain. Dry river channels connected to the floodplain play an important role as sink for flood waters and may make the peak flow decrease (Boulton et al., 2017). In addition, it recharges alluvial aquifers. Higher water levels connect and recharge isolated pools and bring nutrients. In contrast, extreme flood events may damage riparian vegetation, lentic and lotic ecosystems.

In IRES, **erosion and deposition control** depend on the attenuation of runoff and discharge rates. Erosion is mainly controlled by the vegetation and soil erodibility. On the one hand, excessive erosion may cause incision plus the subsequent shrinking of the phreatic level and may damage infrastructure. On the other hand, excessive deposition may increase habitat homogeneity (e.g., filling river pools, so important in IRES), reduce storage capacity of reservoirs, and increase turbidity hence decreasing water quality.

There are no studies about the importance of IRES in terms of **pollination and seed dispersion**. However, we can say that in large agricultural areas, unmanaged vegetation is concentrated in IRES, so it can provide habitat for insects that will then pollinate the crops being grown on adjacent land (e.g., nesting sites for bumblebee queens, in Kells and Goulson, 2003). Moreover, IRES often act as corridors for migration of cattle and wild animals (Sánchez-Montoya et al. 2016), which certainly favours seed dispersion (Figure 5.5).

Disease and pest control basically depend on the riparian habitats' capacity of housing invasive species and pathogen vectors. For instance, pools of IRES during drying phases may be key habitats for mosquitoes that transmit pathogens, whereas drying and flowing phases may avoid their reproduction (Dida et al. 2018). Time synchronization between flow regime and crop and vector's phenology is an important factor for the proliferation of disease and pests; and managers can avoid it by preserving native species, natural flow regimes and good ecological status (Duchet et al. 2017).

5.2.3 Cultural ecosystem services

Cultural services are defined as the non-material benefits people obtain from ecosystems. They usually can have both a direct or indirect use, non-consumptive, and a subjective value. However, an excessive flow of these services can also cause the degradation (e.g., by overcrowding) and commercialisation of nature.

Aesthetic values are the benefits associated with the visual, auditory and olfactory perception of IRES. Aesthetic values are of particular importance as sensory stimulation is one of the most intimate links that people have with ecological phenomena. IRES represent landscapes in which local public has interacted and related in very special ways becoming important landscapes by its visual characteristics. They may attract tourism as well (see the example of Figure 5.6)

The provision of **recreational activities** is presented in multiple ecosystems in very different ways. In IRES, such activities also differ between dry and wet phases. For instance, trekking and hiking are possible when the river runs low or dry, and canyoning, swimming or fishing when water is present (see Figure 5.7). Besides the flow level, recreational services are closely dependent on the weather too. Recreational activities may not only provide direct economic benefit from tourism, but also contribute to the physical and psychological health of people.

Environmental education and scientific knowledge is the capacity of IRES to generate and disseminate socio-ecological knowledge, such as the importance of temporal and spatial variability for IRES or the different uses of dry riverbeds for local societies. Educational and scientific activities promote pro-environment attitudes that can indirectly improve the perception towards IRES and, subsequently, the improvement of the ecosystem health and service provision. See Section 5.5.4 for methodologies to engage with stakeholders.

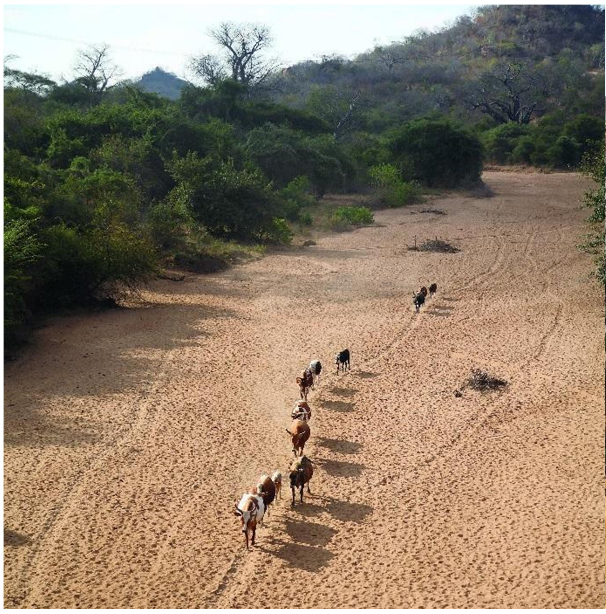

Figure 5.5 Seed dispersal. The use of IRES as passages by shepherds favours seed dispersal in Mozambique.

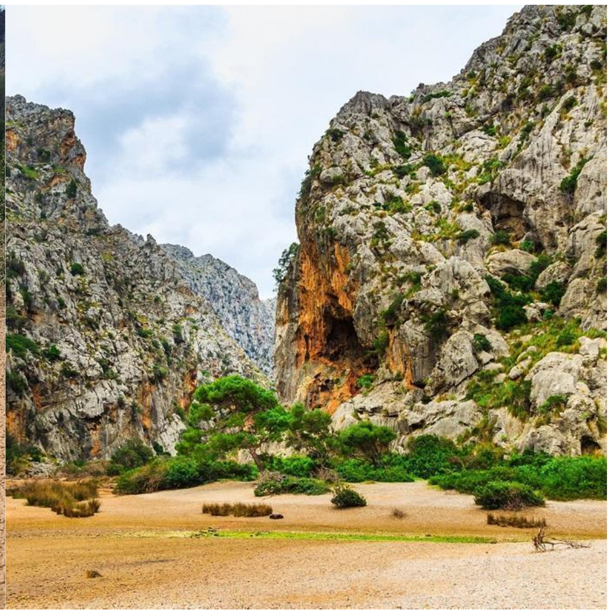

Figure 5.6 Aesthetic values. Torrent de Pareis, Escorca, Mallorca, Spain, is a tourist place for its spectacular scenario.

Figure 5.7 Recreational activities. Canyoning is an activity usually done in small rivers like IRES. This picture is taken in Fischen im Allgäu, Germany.

Figure 5.8 Local ecological knowledge. Pond water crowfoot (Ranunculus peltatus) contributes to the flowing-phase character of winterbourne chalk IRES in the south of England. © Andy House.

Figure 5.9 Local ecological knowledge. Traditional irrigation system (called ‘acequias’) based on the maximization of the profits from an extremely variable flow regime in Sierra Nevada Mountains, Granada, Spain. © Cristina Quintas-Soriano

Figure 5.10 Spiritual and religious services. This is Quema River ford, and the Triana brotherhood on procession to the hamlet of El Rocío, Spain.

Local ecological knowledge is transmitted from one generation to the next when the IRES is well-preserved, and the ecosystem is degraded, and can provide sense of place (Figure 5.8). This service maintains for example the awareness of flash floods, which are very common in IRES, the ancient irrigation systems adapted to the IRES variability (see Figure 5.9), the visibility of those other services usually neglected by the environmental or water administration. With respect to ecotourism, a better local ecological knowledge improves the tourism supply, strengthens the tourism workers’ skills, and offers a wider variety of sustainable leisure activities.

The deficit of knowledge on IRES-specific **spiritual and religious services** reflects the paucity of a wider consideration of cultural ecosystem services. Rivers and springs have attracted people since prehistoric times for perceived physical healing benefits. In many cases, these places were sacred for worship, sanctuary and pilgrimage, as well as spiritual fulfilment. Examples include the shrines to the Virgin Mary like in Fatima (Portugal) or Medjugorje (Bosnia-Herzegovina), and the Chalice Well at Glastonbury (UK), with Celtic origins. Figure 5.10 shows a ford crossing an IRES used in pilgrimages to El Rocío (Spain).

Today, **therapeutic services** are more linked to nature-based health care provision, so-called ‘nature on prescription’ or ecotherapy. In order to assess spiritual, religious and therapeutic services, an interdisciplinary approach is essential: ecologists and social scientists working with stakeholders, such as spiritual and indigenous groups, health care providers, and agencies that facilitate and promote practical interactions with the conservation of rivers.

Finally, the fact that cultural ecosystem services are the most context-dependent of all services, makes particular uses emerge in different contexts and times. Thus, IRES are also used as car parks in populated areas, or even as dumping sites that wait for a flash flood to sweep away the rubbish where the stream is ephemeral.

5.3 Drivers of change of ecosystem service provision

5.3.1 Morphology

The morphological features of watercourses may greatly influence the ability of rivers to provide services. The basic morphometric parameters of the riverbed that directly affect the quality and quantity of service provision include the level of confinement, the channel sinuosity and the riverbed roughness. On the one hand, the level of confinement may explain the quantity of service provision. For instance, the less confinement, the wider the floodplains are and the more timber biomass. On the other hand, the quality is more influenced by the channel pattern (i.e. single or multiple thread), and by the type of substrate (e.g., bedrock, alluvial gravel or silts). The more diversity of morphological features and habitats, the better the provision of gene pool protection is and a diversity of recreational activities. Channel morphology is also one of the main variables that determine drying conditions of the riverbed, which is very related to hydrology and also determines the provision of services.

5.3.2 Hydrology

Hydrological variability characterizes IRES, being one of the most important variables that control not just freshwater provision but most ecosystem services. Ecosystem services provision depends on the aquatic states, as well as on their duration, frequency, timing and intensity. For instance, since greenhouse gases are released during rewetting events, the number of such events highly influences IRES 'role on climate regulation. An increase in the number of zero-flow days can compromise recreational swimming in pools, because isolated pools may not be attractive for swimming after weeks of flow disconnection due to contraction (size reduction), algae development, and decrease in water quality. On the other end, perennialization of IRES would reduce the provision of regulating ecosystem services such as flood and erosion control that are maximized during the dry phase when dry channels act as sinks for floodwaters and sediments.

5.3.3 Biogeochemistry of drying out and rewetting

Intermittence and the dry, wet, and transitional phases strongly influence nutrient inputs, in-stream processes, and downstream transport (see von Schiller et al. 2017 and Chapter 3 of this Handbook for more information). Thus, biogeochemistry drives ecosystem services provision in relation to the regulation of the carbon, nitrogen and phosphorus cycles. Carbon sequestration as well is related to climate regulation, while the release of phosphorus and nitrogen nutrients is important for fishery production downstream, and their retention improves water quality. In some cases, increases in organic matter and nutrient concentrations after rewetting from dry conditions in IRES can cause eutrophication and potentially lead to the occurrence of hypoxic blackwater events (Hladyz et al. 2011). This has not only an effect on the provision of fish and drinking water, but also on the aesthetics, since is not perceived as either visually or olfactory pleasant.

5.3.4 Biological communities in the interphase between the aquatic and the terrestrial

IRES species interact with each other and their environment to deliver cultural, provisioning and regulating services. As IRES shift between flowing, ponded and dry states, lotic, lentic then terrestrial species dominate communities, and service delivery thus changes over time. In all phases, cultural services reflect species' enhancement of recreation. For example, pond water crowfoot contributes to the flowing-phase character of 'winterbourne' IRES in S outh England (Figure 5.8). Provisioning services are most clearly delivered by human consumption of fish during wet phases – and by excavation of aestivating fish during dry phases in arid regions. In addition, desiccation-tolerant organisms may be sources of biochemical products. For example, molecules from a specialist fly larva have informed development of techniques to preserve mammalian tissues prior to medical use. IRES also provide regulating services, for example microbial processing reduces concentrations of inorganic nutrients, including those of anthropogenic origin, and longer water residence times enhance processing of both nitrate and phosphate when flow ceases.

5.3.5 Landscape and human activities

Landscape and human activities interact to one another to provide all types of ecosystem services. Bearing in mind the agricultural landscape, the presence of an IRES may imply the improvement of the quality of the drainage waters, which are rich in nutrients, and may mean habitat for pollinators and for pest predators. IRES running through urban areas may be green spaces for recreation and inspiration, and purify the air and smooth extreme climate events (e.g., heat waves). But IRES can also be isolated and degraded places where people go to dump their rubbish.

5.4 IRES and society

The perception and values of any ecosystem is also very correlated to the efforts of the administration and the society to preserve it. Dialogue and knowledge sharing about IRES helps improve people's perceptions and strengthen the values upheld, which is very important for the preservation of IRES, as well as for the prevention of related conflicts.

5.4.1 Management issues, trade-offs and conflicts

In environmental management, trade-offs are more likely to occur than win-win solutions. By identifying trade-offs, we can acknowledge diverse interests in managing IRES, detect inequalities in the distribution of ecosystem services benefits and prevent conflicts. Different types of trade-offs can identify different relations to IRES and to which managers should pay attention:

- Social trade-offs (between social classes, ethnic groups, or gender). For instance, in many places of Southern Europe, women usually do not participate from water governance in irrigated landscapes although they may work as farmers and benefit and use IRES (Molina et al. 2006).

- Inter-stakeholder trade-offs. For example, canyoning may be incompatible with native crayfish habitat, hence with the conservationists' will. And irrigators, recreationists and environmentalists may differ in their optimal management of flow regimes (see Jorda-Capdevila et al, 2015, and also Chapter 6 of this handbook).
- Spatial trade-offs. For example, the use of fertilizers and pesticides in crops in the river floodplains influence fish health and consequently anglers in the river channel. Another example, a dam upstream has an impact on all other uses downstream.
- Temporal trade-offs. For example, the trade-off between one generation that over-exploits the river by extracting gravel, and the next generation, which receives a degraded river.

When trade-offs exacerbate an impact, and a social group perceives that it has been neglected, its rights denied or its interests reduced, a conflict may appear. Environmental conflicts usually face two different types of groups that are distinguishable because they show opposed management solutions. Watershed authorities should not only pay attention to their positions – usually difficult to merge –, but also on their interests and needs. Often it is easier to bring the stakeholders together into a third solution. Other important aspects in a conflict are the influence levels of different stakeholder groups and the type of interest they have, for instance, a broad interest in terms of the diversity of ecosystem services that they benefit from versus a narrow interest, or an individual versus a collective interest. Typical environmental conflicts that concern IRES are related to land uses (see Box 5.1) or water management (Jorda-Capdevila et al., submitted).

Box 5.1. Example of a land use-related conflict in Menorca, Spain.

For centuries, the agricultural fields of the island of Menorca (Spain) have been delimited by dry stone walls (delimiting *tanques* – fields) and by drainage ditches (delimiting *daus* – smaller areas within the fields). The function of these traditional ditches was to improve the drainage of the fields in case of heavy rain, preventing flooding of the crops. These lead to major ditches, small canals or streams, thus constituting the first level of the hydrological networks of the Menorcan drainage basins. Over the last decades these ditches have been removed in the fields closer to streams, where the terrain is flatter, and the yield of agricultural work (plowing, sowing, harvest) can be easily improved by using larger machines. Over the years this has led to farmers complaining to the administration claiming that streams full of natural vegetation prevented the proper drainage of water and thus flooded the fields. This has resulted in (i) an increase in the frequency of mechanical cleaning of the streams with heavy machinery, eliminating all the natural vegetation without distinction and (ii) an increase in erosion.

Comparison of the same agricultural fields on 1956 and 2010, where it is possible to see the removal of the drainage ditches. Images from IDE Menorca.

5.4.2 Social perceptions and values

There is a vague appreciation of IRES by the public, which affects not only biodiversity and their ecological interiority but also the variety of ecosystem services they provide to people (Koundouri et al. 2017). Factors that define the disconnection between people and IRES are diverse and depend on cultural roots and socioeconomic context. For instance, in many Mediterranean regions commonly known as ‘ramblas’, there exist an aversion to IRES because they are perceived by the public as dangerous areas or used for as convenient dumping grounds for rubbish (Castro et al. 2019), therefore ignoring the fundamental role they play in preserving key services such as flash-flooding control or groundwater regulation (Armstrong et al. 2012).

Moreover, there is a bias related to the management and policy domains across multiple scales, which, influenced by the rapid need to meet society's ’ needs (i.e., urbanization and agricultural expansion), have been unable to ensure sustainable management and conservation strategies of IRES. Moreover, the failure of capturing a plurality of values associated to IRES is largely responsible for the widespread environmental degradation of these ecosystems (Boulton, 2014):

- Traditional assessments of ecosystem services have been mainly focused in valuing the use values or **instrumental values** of ES, e.g., fishing and birdwatching.
- **Intrinsic values** are also important to be considered. They represent the value that IRES have in themselves and are usually associated to, for instance, aesthetic value or sense of place.
- Many conservation concerns and conflicts could be better understood adding a third group of values called **relational values**, which can be defined as the social preferences, human principles, and virtues that articulate individual and collective relationships between humans and IRES. Relational values of IRES are related to cultural identity, social cohesion or nature stewardship.

5.5 Methods for assessing the value of ecosystem services

A recent report on rivers and streams assessment coordinated by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services highlights that the valuation of ES has focused almost solely on perennial rivers, streams and reservoirs (Castro et al. 2016a,b), however, the value of ES provided by intermittent rivers and ephemeral streams have been largely overlooked (Koundouri et al. 2017, Boulton, 2014).

5.5.1 Indicators for service supply and demand

The ecosystem service concept constitutes new approaches in which it is possible to understand the linkages between ecosystems and social systems. In this sense, an ecosystem's capacity to provide services (supply side) and their social demand (demand side) highlights that the status of an ecosystem service is influenced not only by the ecosystem's properties but also by societal needs (Castro et al. 2013). Burkhard et al. (2012) defined the supply side as the capacity of a particular area to provide a specific bundle of ecosystem services within a given time period, and the demand side as the sum of all ecosystem services currently consumed, used, or valued in a particular area over a given time period.

Table 5.1 shows a variety of indicators which can be used to estimate supply and demand of specific ecosystem services. Most of them are very general, applicable to many other kinds of ecosystems. In order to adapt them to IRES, it is important to consider the temporal variability of their flow regime. A monthly calculation of the following indicators may be enough to integrate dry, pool and flowing phases – when necessary – in any assessment of ecosystem services provision.

Table 5.1 Supply and demand indicators for the ecosystem services of IRES.

Ecosystem services	Indicators of supply	Indicators of demand
General indicators		Social perception / importance perceived
Provisioning ecosystem services		

Freshwater provision	Water yield	Water consumption
Food provision	Population size of species of interest (fish and waterfowl) Richness, abundance and distribution of wild riparian species that provide fruits or grains Fertile area within the river	Number of fish/hunting licenses Game quotas Fish catch rates Crop production Consumption rates of fish/game stocks/wild fruit and grains/crops
Provision of raw materials	Richness, abundance and distribution of wild riparian species that provide fibre and fuel Rates of sediment accumulation	Weight of extracted fibre and fuel Weight of extracted sediment
Provision of genetic resources	Biodiversity indices	No specific methods known. General methods can be applied
Regulating ecosystem services		
Climate regulation	Fluxes of POC and CO2 Presence of woody debris	No specific methods known. General methods can be applied
Air quality regulation	Riparian forest cover	No specific methods known. General methods can be applied
Nutrient cycling regulation	Presence of geomorphological elements, aquatic plants and biofilms, and riparian forest Flow regime Drying-rewetting oscillations	Water consumption production of wastewater
Regulation of water flow and protection against extreme events	Groundwater recharge Area of unconstructed floodplain Capacity of dam storage	Population living there
Erosion and deposition control	Vegetation cover Number of sediment tracks	No specific methods known. General methods can be applied
Pollination and seed dispersion	Quality of river habitat (e.g., MiQu, GUADALMED and ECOBILL protocols) Rate of flower visitations by aquatic insects Pollination Suitability Index for Riverine Landscapes	Area of crops that need pollination surrounding the stream Number of mammals using the stream as corridor
Disease and pest control	Abundance of mosquitoes able to transmit vector	Population living in or visiting the surroundings
Cultural ecosystem services		
Aesthetic values	Number of “viewer days” per year or the monetary value of a change in scenic quality	Population living there Number of visitors/tourists Pictures posted in social networks
Space for recreational activities	Very variable depending on the activity Travel cost method (assessing variations in travel effort across visitors)	Population living there Number of visitors/tourists
Education and research	Spatial models	Research papers with the IRES as case study Visits by schools / student surveys
Local ecological knowledge	Richness of profitable or iconic species Knowledge on the dynamics of the ecosystem flora and fauna of symbolic, mythic or totemic significance	Population living there Social media analysis Residents surveys
Spiritual, religious and therapeutic services	Presence and extent of protected areas, sacred/religious sites, pilgrimages, festivals or rituals, folk songs, myths, legends or genealogies	Population living there Number of visitors/tourists Mental health and wellbeing-related metrics

5.5.2 Monetary valuation methods

Monetary valuation methods aim to express the total economic value of an ecosystem in monetary terms. Monetary valuation methods are sometimes criticized because of the risk of commodifying nature for its own conservation. However, their main advantage is that they make Nature's values visible. Since many ecosystems do not have a market price, their values are often overlooked in the decision-making processes. This can lead to environmentally damaging practices. By estimating the value of ecosystems, monetary valuation methods highlight their importance and ensure their benefits are incorporated in public decision-making. This is ultimately expected to lead to a sounder management of natural resources. Monetary valuation methods have also been found to be very useful for the internalisation of environmental externalities, which may be done through environmental pricing (e.g., green taxes, subsidies for environmentally friendly practices) or inter-stakeholder negotiations (e.g., payment for ecosystem services). See some information on the implementation of an economic valuation for an efficient environmental management in Box 5.2.

There are many different methods for the monetary valuation of ecosystems. Those that might be most useful for environmental managers are described thereafter. For more detailed information on each method, we recommend Chapter 5 in "The Economics of Ecosystems and Biodiversity" (Pascual et al. 2010).

If a product of IRES that is valued has a market price (e.g., fish sold at the market), then the quantity (e.g. fish in kg) produced by IRES can be multiplied by its price. However, most aspects of IRES ecosystems do not have a market price. In those cases, one of the following methods can be applied.

The **hedonic pricing method** is appropriate when IRES is located relatively close to human settlements and might affect (i.e. increase) the housing prices. This method estimates to which extent the presence of IRES explains variations in housing prices.

The **travel cost method** is designed for the valuation of those ecosystems that are used for recreational activities. Travel expenses and travel time spent for visiting an IRES represent the "price" visitors are willing to pay for accessing IRES. The method is most suitable for touristic IRES, such as the Torrent de Pareis (Figure 5-6).

The **contingent valuation method** directly asks respondents in the survey how much they would be willing to pay for an ecosystem or a change in its quality or quantity. It is dependent on the hypothetical scenario describing the change in the ecosystem.

The **choice experiment method** estimates willingness to pay based on the choices and trade-offs that respondents make in the survey between two or more hypothetical future scenarios. Each scenario is described by a number of characteristics or attributes. The method allows to value different attributes of the same ecosystem, for example, the number of aquatic bird species, water quality and the aesthetic aspect of IRES.

Finally, the **benefit transfer method** takes the monetary value of an IRES from another case study, ideally with similar biophysical characteristics and socio-economic context. This method is advisable only when resources (time, money or personnel) for collecting own data are limited and monetary values for other case studies exist, which is not yet the case for IRES.

Box. 5.2. Implementation of an economic valuation for an efficient environmental management.

Water is a social but also an economic good for which it is important to identify and define its services and uses as well as the costs related to water use, i.e. financial/supply costs, environmental and resource costs. As discussed in Koundouri (2015) and Koundouri et al. (2019), IRES and water resources remain a public good. Thus, provision to one individual does not prevent others from using it. This is a form of market failure and can result in misallocation of resources. With regard to water quality, excessive pollution is caused by the existence of environmental externalities (e.g. waste treatment plants, factories, urban and agricultural run-off). Government failures can also lead to misallocation of resources, as for example subsidies for agricultural production leading to the overexploitation of water resources for irrigation purposes. As a result of these market inefficiencies and externalities, the natural resource is not allocated efficiently among alternative resource users. Allocative efficiency requires the identification and monetisation of the resource costs (i.e. the foregone opportunities which other uses suffer due to the depletion of the resource beyond its natural rate of recharge or recovery) and environmental costs (damage that water uses impose on the environment and ecosystems and those who use the environment). The economic literature proposes several monetary approaches to estimating these costs. Nevertheless, the quantitative findings remain sporadic while the transferability of the results from one site to another may be subject to limitations.

5.5.3 Non-monetary valuation methods

The benefits that people derive from ecosystem services provided by IRES can be assessed with non-monetary methods too, called also as socio-cultural methods. These include both individual and group-based methods, where dialogue with experts or resource users – either beneficiaries or losers – can reveal how they perceive IRES, what importance they attribute to it, and what benefits they realize in different localities and periods of the year. Socio-cultural methods are able to reveal a wide range of values, from intrinsic (ecological) to relational (social) and instrumental (economic) values, and especially well-suited to understand and characterize intangible benefits that cannot be measured quantitatively (e.g. values of cultural ecosystem services). Visiting some hot spots or favourite places with respondents or using pictures as proxies – as suggested by photo-elicitation studies, photo-series analysis, photo-based Q-method, or the photovoice method – can help better characterize the variability of benefits according to the different phases of the river, which is crucial for the valuation of IRES. While socio-cultural methods do not monetize the value of ecosystem services provided, quantification is possible e.g. via simple ranking or scoring exercises, or by the collection of numerically available data (e.g. quantities of harvestable fish or the number of issued fishing licences), which can be further visualized in multi-layered maps. See more information on non-monetary valuation methods in Santos-Martín et al. (2016).

5.5.4 Engaging the beneficiaries

It is increasingly recognized that public participation is instrumental in laying the groundwork for sustainable practices in physical planning and management as well as social community building. In fact, as argued by specialists from different domains, to achieve sustainable communities it is necessary to: 1) involve local citizens, 2) allow citizens to analyze their own problems and fashion their own solutions, and 3) support community initiatives which allow them to be the instruments of their own change. Attention to sustainable community development practices foster social goals which can strengthen the connections between participatory practices and government or authority decision making. Of course, there are several participatory levels, which go from passive participation to active participation (see Figure 5.11).

<table>
<tr><td rowspan="7">What role do you play now as a citizen?</td><td>Active Participation</td><td rowspan="7">What role would you like to or think you should play as a citizen?</td></tr>
<tr><td>Citizen as Decision maker: Citizens of a community have the clearest and perhaps the most accurate perception of needs and priorities of their community and should make the decisions themselves.</td></tr>
<tr><td>Citizen as Consultant: Citizens should occasionally be consulted to contribute their professional opinions during the decision-making process, and when given adequate information can make educated decisions about various proposals.</td></tr>
<tr><td>Citizen as Respondent: Citizens do not necessarily know what is needed or what is the best approach, but their opinions should be surveyed and analyzed by well-trained experts and used in the decision-making process.</td></tr>
<tr><td>Citizen as Constituent: Expends on trained elected representatives have the right to make decisions on behalf of citizens and to assume that they are representing their constituents'' interests unless hearing otherwise.</td></tr>
<tr><td>Citizen as Voter: Citizens should vote for their representatives, but public decision making is a scientific pursuit and should be left to skilled experts and policymakers, not the general public.</td></tr>
<tr><td>Passive Participation</td></tr>
</table>

Figure 5.11 The role of the citizen in the decision-making process. Source: Regional Environmental Centre for Central and Eastern Europe - REC (1996).

In this subsection we summarize three different methodologies of public engagement for the gathering of social data on social preferences about ecosystem services in IRES: participatory mapping, citizen science, and participatory scenario planning.

Participatory mapping is a non-monetary valuation method that seeks the spatial relation between landscape characteristics and human wellbeing. By engaging the general public and stakeholders to identify place-based local knowledge, the method contributes to quantify supply and demand of provided services. This can be a facilitator for decision-making and communication. In IRES, to our knowledge so far, no exercise has been done on participatory mapping of ecosystem services. One study performed in the Ter River basin (Spain) analysed the perception of the local people and water administration about a small dam, and the expected perception about its removal. In the study, different interviewees were asked to draw supply and demand sites of perceived ecosystem services and their level of importance from 1 to 5 (Brummer et al. 2017).

Citizen science (CS) nowadays is defined as any practice of public participation and collaboration in scientific research. Although, in a more classical definition of CS, the public participation focuses mostly in data collection, especially for CS projects born from disciplines like biology, ecology, environmental sciences or hydrology. Thereby, CS projects focused on rivers usually ask for data about water quality or quantity, and some of them apply simplified bioassessment methods. On the other hand, very few projects ask citizens about their perception of the fluvial ecosystem or, directly about ecosystem services. Moreover, they are almost nonexistent for IRES, even though the collected data about ecosystem services might be useful to enhance participation and empower people in future management (see an example in Figure 5.12).

Figure 5.12 RiuNet (www.riunet.net) is a Citizen Science (CS) Project that allows citizens to assess the hydrological and ecological status of IRES, as well as to inform about their cultural and social values such as bathing, aquatic sports, fishing, hiking, research and educational, aesthetics or inspirational values (see chapter 2, section 2.3.3).

Participatory scenario planning can be applied in ecosystem services assessments to collect social perceptions and initiate public dialogue about the benefits and values attached to certain ecosystems. If applied in a group-based format, scenario planning can involve various stakeholders, experts or citizens. Scenario planning starts with 1) identifying the major drivers (either socio-political or ecological ones) that influence the future state of a given ecosystem and 2) assessing the current state of the ecosystem. Based on the drivers and the current status, 3) alternative scenarios can be developed for the future, and then 4) scenarios can be evaluated in terms of how the ecosystem and its services will change, and how human well-being will be impacted. The public dialogue around the scenarios does not only allow us to understand which ecosystem services are of priority and why, but also helps local communities to plan future actions to preserve crucial ecosystem services. Participatory scenario planning is widely used in mixed ecosystems, although not many examples are known directly for IRES. In the OpenNESS project participatory scenario planning was applied in an area in central Hungary with temporal alkali lakes mosaics with open grasslands and forest steppes.

5.6 Conclusions

5.6.1 Take-home messages

- There is an urgent need to understand the different world views, ways of knowing, and public attitudes and perceptions that form the basis of values towards IRES. Incorporating these values into transdisciplinary processes will allow decision makers to address conflicts over IRES and enhance public perceptions and values regarding IRES. The ecosystem services concept may be useful to incorporate those values and understand the relationship between IRES and society.
- IRES can be of natural (due to factors related to the climate, the geology or the size of the catchment) or anthropogenic origin (e.g. perennial rivers that become IRES as consequence of flow regulation and water abstraction). This should be considered when assessing the provision of ecosystem services since they would require different approaches, assessment criteria and reference values which would ultimately determine a positive or negative overall evaluation.
- IRES may provide a wide spectrum of ecosystem services, e.g., by providing cultural services in Mediterranean climates and the habitat of iconic species in chalk streams, by being a corridor for people, livestock and wild animals especially in arid and or anthropized environments, and by providing goods such as water, sand, edible plants or timber.
- Socio-cultural values of IRES change over time, not only among aquatic states, but also among seasons and years. Their value is not only intrinsic to type of ecosystem either, but depends on the sociocultural context, which may also change. Thus, by improving the biophysical condition, knowledge and awareness about IRES, managers can modify the way in which society perceives and values IRES.
- The economic literature develops a set of methodologies and approaches that can be implemented with regard to monetizing the values of ecosystem services provided by IRES. Policy makers and managers need to consider these alternatives to develop an optimal approach to efficient environmental management. In doing so they also need to consider: i) the full spectrum of multiple pressures put on river bodies and water supply, ii) the full range of users and beneficiaries, iii) the “polluter pays” principle and the fair allocation of cost recovery among different users and, iv) affordability and competitiveness of full cost recovery of water services.
- Non-monetary techniques for the assessment of ecosystem services are necessary to integrate instrumental, intrinsic and relational values associated with IRES. Through the engagement of experts and/or resource users, these techniques can reveal people’s perceptions towards IRES, giving voice to intangible and experiential benefits from IRES, hence making them more suitable for mutual learning processes, balancing inter-stakeholder conflicts and optimising management for socio-cultural, economic and environmental outcomes.

5.6.2 Future directions in ecosystem services of IRES and recommendations for managers

The greatest milestones of nature’s conservation are being achieved thanks to, not only researchers and managers, but to an empowered civil society that steps up when necessary. It is the society that has a positive conception of their rivers and cares about their rivers, even if they naturally run dry. It is often said that IRES are less appreciated by the local public than perennial rivers, but they are actually less appreciated by researchers, policy makers and managers too (Acuña et al., 2014). Thus, there are two main challenges to work on in the future. First, we should acknowledge those socio-cultural values associated with IRES that are usually neglected but empower people to become actively involved in decision-making about their local environment. This can only be done by broadening our perspectives towards a more interdisciplinary management that includes instruments from, e.g., ethnoecology, environmental psychology or ecological economics. Second, we should be able to raise awareness of social-ecological values of IRES, especially in those cases when the local people live with their back to the river. Environmental education is key here. However, it is more important is that managers pay attention to IRES with the same will that they do to perennial rivers, for instance, by incorporating them in the water management plans and policy.

6. Environmental flows: assessment and implementation in IRES

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6.1 Introduction

- In this chapter, the term "Eflows" is used to both represent "environmental flows" and "ecological flows". The revised definition of "environmental flows" according to the Brisbane Declaration (2007, 2018) is as follows: "*Environmental flows represent the quantity, timing, and quality of freshwater flows and levels necessary to sustain aquatic ecosystems which, in turn, support human cultures, economies, sustainable livelihoods, and well-being".* In the definition, we understand that maintaining the required flow for ecosystems will also provide socioeconomic benefits as described in Chapter 5. But we also want to refer to the term "ecological flows" defined by the European Commission in the following paragraph.
- According to the Guidance Document No. 31 of the implementation of the Water Framework Directive (WFD) (EC, 2015), ecological flows or "Eflows" are considered within the context of the WFD as "an hydrological regime consistent with the achievement of the environmental objectives of the WFD in natural surface water bodies as mentioned in Article 4(1)". Considering Article 4(1) of the WFD, the environmental objectives refer among others to:
    - Non-deterioration of the existing status or achievement of good ecological status in natural surface water bodies. Therefore, suitable flow regime (also called Eflows) have to be preserved or applied in order to meet good or high ecological status considering all biological quality elements set by the WFD;
    - concerning the hydromorphological elements, they represent support quality elements and are only used to distinguish good ecological status from excellent or high ecological status.
    - Compliance with standards and objectives for protected areas, including the ones designated for the protection of habitats and species where the maintenance or improvement of water status is an important factor for their protection, including relevant Natura 2000 sites designated under the Birds Directive (BD) and Habitats Directives (HD). Those habitats or species can require additional or specific flows which must be considered by water managers;
- There are more than 200 methods for determining Eflows, but a few of them are compatible with IRES;
- In order to value IRES and design and implement Eflows for IRES, eco-hydrological relationships must be studied beyond the flowing period and include also the "pool" period;
- Implementation of Eflows can take place via valuation of ecosystem services and/or included in future allocation of water rights;

- Some heavily modified rivers might never reach the good ecological status required by the WFD;
- To reach good ecological status in the EU, each country must perform a cost-benefit analysis to find the best way to implement Eflows while satisfying economic and social needs.

6.2 Eflows in IRES for the ecosystem integrity and the provision of ecosystem services

6.2.1 The role of the flow regime on the IRES ecosystems

Temporal variability of water discharge (i.e., flow hydrograph and related flow characteristics) has one of the most profound impacts - along with the sediment regime and vegetation dynamics - on the structure and function of lotic[1] ecosystems. What is true for perennial streams, is also true for IRES (see chapter 4). In IRES, the recession of flow is as important as its onset. Recession of flow transforms lotic habitat to lentic[2] habitat, thus forcing ecological communities to adapt or perish. Timing of flow recession is a key factor in determining the fate of newly created pools. If recession happens too early, pools might dry before a life cycle has been completed. Flow magnitude is also important in IRES. Local scouring in river beds during high flow stages is essential for creating and maintaining pools, flushing downstream the accumulated finer sediments. Sediment transport creates new lotic habitats but may also bury or disconnect existing ones. More information about the role of flow regime are found in Chapter 2 in the book.

Vegetation encroaching riverbeds is common in IRES. Low shear stress[3] will result in streams encroached with woody vegetation, while strong shear stress will result in a more open stream with grasses and annual vegetation. In Mediterranean IRES, the end of the harsh and flashy “wet season” creates a window of opportunity for growth in benign environmental conditions. Conditions are set to deteriorate until partial rejuvenation takes place in the next “wet season”. Unpredictability of flow may also affect IRES communities such as unexpected, out of season droughts or flash floods.

6.2.2 The importance of sediment transport for Eflows

Attempts to investigate the current trends in river sediment loads face the problem of data scarcity, despite the evidence of recent and ongoing river incision due to sediment starvation in European rivers (Comiti and Scorpio, 2019). Indeed, measurements of sediment fluxes are available for very few rivers worldwide, and in most cases only the suspended sediment load is monitored while measurements of bed load transport are lacking. Also, meaningful analysis of temporal trends in annual sediment loads requires records of appreciable duration but long-term sediment monitoring programs are even fewer (see e.g. Rainato et al., 2017).

Suspended load accounts for the vast majority of the total sediment load: in most of the monitored rivers it was found to be in the range 80-95% (Walling and Fang, 2003; Rainato

[1] Lotic habitat represents the habitat in flowing water such as rivers.
[2] Lentic habitat represents the habitat in stagnant waters such as lakes.
[3] Shear Stress is a measure of the **force** of friction from a fluid acting on a body in the path of that fluid. In the case of open channel flow, it is the **force** of moving water against the bed of the channel.

et al., 2017), with the lower values measured in steep mountain creeks. Nonetheless, although bedload transport comprises a minor fraction of the total sediment load, it is by far the main driver for the morphological dynamics of stream beds, and thus its quantitative knowledge is key to predict ecological changes associated with flow variations. In fact, a quantitative description of the sediment regime of a specific stream is a fundamental step towards the understanding of its ecological dynamics (Wohl et al., 2015).

From a geomorphological perspective, the land-ocean transfer of sediment by rivers is also a key component of the global denudation system. Knowledge of the amounts and dynamics of suspended sediment in river systems is important, as both the flow regime and in-stream hydrodynamics are fundamental controls on the transport process (Van Rijn, 1984). Regarding IRES, the transport of sediments during the rewetting phase was shown by Shumilova et al. (2019) to drive the most relevant flux of nutrients and DOM.

6.2.3 Environmental flows for ecosystem services

Eflows describe the quantity, timing, and quality of freshwater flows and levels necessary to sustain aquatic ecosystems (so achieving or sustaining good ecological status in water bodies according to the WFD) which, in turn, support human cultures, economies, sustainable livelihoods, and well-being (Arthington et al. 2018). Additionally, the Brisbane Declaration (2007) not only defines e-flows as essential for freshwater ecosystems' health, but also for the preservation of human livelihoods. This implies that freshwater ecosystems substantially contribute to human well-being, and that the preservation of flow regimes are indispensable to preserve both ecosystems and livelihoods (Acreman *et al.*, 2014).

IRES, considerably widespread throughout the globe, are not less important than permanent rivers, and provide multiple benefits to our societies, ranging from cultural to regulating or provisioning ecosystem services (see Chapter 5). Thus, the implementation of Eflows that may change the duration and frequency of wet and dry phases has for instance an effect on nutrient cycling and water quality (see Chapter 3), and on the life cycle of specific taxa that might be iconic in a specific socio-cultural context or extracted for a specific usage (see Chapter 4). Certainly, different components of the flow regime affect different ecosystem services. While the frequency of high flows dictate vegetation density and the associated effects in terms of flood flow resistance, the timing of the zero flow periods is important for the river as a corridor for both wild animals and cattle. Of course not all ecosystem services have the same requirements in terms of flow regimes, neither have to be in accordance with the ecosystem needs. It is indeed a duty of managers to understand and deal with the existing trade-offs that may appear among beneficiaries – and between beneficiaries and the ecosystem – and propose an Eflow regime that better suits their political will.

6.3 Management of flows in IRES

As already stated in Section 2.5.1 above, for assessing the degree of hydrologic alteration in fluvial water bodies and designing the appropriate Eflows, it is necessary to follow four steps (Poff et al, 2010):

i   Find the ‘baseline’ or reference unimpacted regime characteristics for the water body under study,
ii. Classify the stream regime using ecologically relevant variables,
iii. Determine the deviation of the current regime from the baseline-condition one, using ecologically relevant variables as indicators
iv. Develop regime alteration-ecological response relationships.

The three four steps have been discussed above in sections 2.2.1 and 2.4, whereas the three last steps are addressed within this section.

6.3.1 Hydrological alterations in IRES

The ambition of implementing Eflows in the context of the WFD is to improve the status of a river to reach “good ecological status” if it is a natural river or “good ecological potential” if it is a heavily modified river. The assessment of the hydrological regime of a river is of the greatest importance as it allows to assess the divergence of the actual hydrological regime from its ‘natural’ conditions (De Girolamo et al., 2015, Gallart et al., 2012). For any river, natural flow regime components including specific magnitude, timing, frequency, duration and rate of change are the main drivers of freshwater ecosystems and riparian vegetation status (Poff and Zimmerman, 2010), along with the analogous components forming the sediment and the wood regimes in rivers (Wohl et al., 2015; Wohl et al., 2019). For temporary rivers, the interruption of flow and the drying of the river bed are the main hydrological drivers of their ecology (Acuña et al.2017), so their statistics must be at the forefront when their regime is analysed (see Section 2.4).

In a recent study, an inventory of global rivers reports that only 37 percent of rivers longer than 1,000 kilometers remain free flowing over their entire length and 23 percent flow uninterrupted to the ocean (Grill et al., 2019). On a global scale, fragmentation by dams and reservoirs is the main cause of flow alteration and loss of river connectivity (Vorosmarty et al., 2010). Recent studies show that water abstraction for irrigation and future impact of global change will likely put more pressure on freshwater resources in particular via food demand increase and virtual water trade (Pastor et al., 2019; Vörösmarty et al., 2015). More attention has been brought to the relationship between ecology and hydrology of perennial rivers than intermittent rivers (Acuña et al., 2017). However, during the recent decade, a number of studies highlighted the issue that the maintenance of intermittent rivers are fundamental for freshwater ecosystem survival as being hosts to many endemic and specialized species (Acuña et al., 2014; Datry et al., 2018; Gallart et al., 2017). Nowadays, a couple of effects of flow alterations have been identified, the most prominent being water storage with a total capacity of 7200 km^3 from dams and reservoirs (Naiman and Dudgeon, 2011). Water abstraction for agriculture is the most important driver of hydrological alteration which represents 70% of total withdrawals (FAOSTAT). Besides, the return flows from irrigated areas changes the seasonality of IRES with providing water flow to the river during the dry season. Other human pressures such as household and industry consumption, point source discharges such as the wastewater treatment plants (WWTPs), hydropower and land use change can also cause strong hydrological regime alterations affecting the ecosystem, which may lead to its collapse. Other kinds of river alterations exist such as groundwater abstraction which can heavily impair intermittent streams (Karaouzas et al., 2018).

Relevant indicators

Until recently, the hydrology and ecology of temporary rivers have been investigated and assessed following methods designed for permanent rivers (Larned et al., 2010, Acuña et al., 2016). One of the main issues using these methods is the lack of consideration of the pools phase that may occur after the cessation of flow; these pools may last for many months, so that the active aquatic life in lentic habitats may be by far not limited to the duration of flow. But this is not only a conceptual issue but also a methodological one, because the ordinary source of hydrological information, flow records or simulations, do not inform on the possible occurrence of water pools after the cessation of flow.

It may be argued that it is not possible to manage the occurrence of isolated pools by setting Eflows. This is only partially true, because most of the river pools are fed by stream flow that may or may not be measured by gauging stations (Zimmer et al., 2020). Completely isolated pools are infrequent, while those fed by water flowing within the river alluvium or fed by underground waters are the most frequent (Bourke et al., 2020).

At present, by far the more consolidated method for investigating the hydrological alteration of rivers is the Indicators of Hydrological Alteration (IHA), developed by The Nature Conservancy (2009) for permanent or nearly permanent rivers, using recorded or simulated hydrographs. More recently, emerging methods particularly designed for temporary rivers have been developed by de Girolamo et al. (2015) and D'Ambrosio et al (2017) using hydrographs and Gallart et al., (2017) using hydrographs along with other diverse kinds of information that specify the occurrence of water pools, as described in Section 2.3.

The IHA is composed of 33 indicators able to describe the magnitude, timing, frequency, duration and rate of change of flow regime, many of them synthesized in table 2.1. Among the IHAs that have been indicated for characterizing intermittency and flow alteration: zeroflow (ZF) is considered to be the most relevant indicator regulating the aquatic fauna in a temporary stream. However, flow permanence and predictability have been recognized as fundamental indicators to characterize and classify flow regime in intermittent streams (De Girolamo et al., 2015; Gallart et al., 2012). D'Ambrosio et al., (2017) identified the most significant indicators to differentiate the flow regime of rivers in Southern Italy, such as the monthly mean flow of January, March, November and June, the number of days with no flow, the annual maximum 30 and 90-day duration (characterizing magnitude and duration), predictability of the dry season, high and low pulse count and finally flashiness index (characterizing rate and frequency of water conditions changes). These indexes need to be tested/adapted to other watersheds of the Mediterranean.

Using a procedure designed for temporary rivers that takes into account the occurrence of water pools using the six metrics of Table 2.2 obtained from diverse kinds of information, the free software tool TREHS (Gallart et al., 2017) calculates the degree of hydrological alteration using an expert scoring method from the differences between the metrics obtained for the reference regime and the actual one, provisionally assumed of ecological relevance. These calculations are made on a separate auxiliary spreadsheet that can be inspected by the user in order to monitor the process and, if need be, update some interim expert criteria.

First, for every metric, the average and standard deviation of the values obtained from the diverse types of information (hydrographs, interviews and direct or photographic observations) on the reference and actual regime are obtained. The user can switch off any types of information if bias is suspected. Then, the differences between the reference and the available actual metrics are compared with threshold values that depend on the reference Mf value, to decide whether the divergences are acceptable or not; The more permanent the regime, then the lower the divergence of the metrics is permitted.

The criteria used for assessing the hydrological alteration are as follows:

- Decrease of flow permanence Mf, two levels of severity (gentle and harsh).
- Decrease of surface water permanence (Mf + Mp), two levels of severity.
- Increase of flow permanence Mf, two levels of severity.

– Change of seasonal predictability Sd6.
– Change of seasonal patterns SWs or ESs.

Note that an increase in flow permanence or a change in the temporal pattern is also taken as hydrological alteration since they change natural conditions and may facilitate the colonisation of invasive species, particularly fishes and crayfishes (e.g. Riley et al., 2005).

Every criterion is penalised with one negative score that is subtracted from a value of 4; then the Hydrological Score (HS) is determined as ‘not altered’, ‘slightly altered’, ‘moderately altered’ or ‘highly altered’ for total values from 4 to 1 respectively. TREHS displays the criteria used for this determination to inform the manager of the measures needed for regime reclamation. Finally, it also estimates the degree of confidence of the diagnosis, based on the ratio between the metric differences and their standard deviations, as well as its robustness, based on the number of different kinds of information used.

Classification of river regime

It is important that the river regime shift is identified at the earliest stage of a river assessment. Therefore, understanding and classifications of different river regimes are necessary before a shift can be identified.

In section 2.4 the regime classification of temporary rivers is examined and two examples of operational classifications are shown: the prescribed classification operating in Spain using the no-flow days of flow series simulated as a natural regime with the help of a rainfall-runoff model (Table 2.5) and the classification proposed using the statistics of the three main aquatic phases; flow, isolated pools and dry river bed (Figure 2.5 and table 2.6). The former is more easy to use but less ecologically sound and useful than the latter.

Further to these two approaches, another operational classification first proposed by Gallart et al. (2012) and further developed by De Girolamo et al. (2015) is briefly shown here because of its potential interest for hydrologic alteration assessment and Eflows design (Table 6-1). This classification uses only observed and simulated flow series but takes into account not only the flow permanence but also the seasonality or predictability of the no-flow period (*Mf* and *Sd6* respectively in Table 2.2). The degree of hydrologic alteration may be easily assessed using the differences of classes and metrics between the natural and the actual regimes (De Girolamo et al. (2015). The reference made there to the occurrence of pools is not based on observations, but on assumptions derived from the combined flow permanence and predictability metrics.

Table 6.1 Classification of stream type (adapted from De Girolamo et al., 2015 and Gallart et al., 2012)

Stream type	Flow duration (month.year^{-1})	Pools duration (month.year^{-1})	Dry period (month.year^{-1})
Permanent (P)	≥ 10	≤ 2	No occurrence
Temporary intermittent (IP)	≥ 3	≤ 9	No occurrence
Temporary intermittent dry (ID)	≥ 3	≥ 1	≥ 1
Ephemeral (E)	≤ 2	Variable	≥ 10

6.3.2. Impacts on ecosystems

While ecosystems of intermittent streams tend to be more adapted to drying than those of perennial streams, a recent study shows that hydrological disturbance was more pronounced in the intermittent streams of the Evrotas River catchment than in the perennial streams concerning species density and percentage composition (Skoulikidis et al., 2011). The major impact of flow alteration is a reduction in the amount of necessary flow for ecosystem maintenance and/or survival. However, flow alteration can also lead to ecosystem collapse and/or shift by changing flow seasonality due to dams and reservoirs flow releases during the dry season (Datry et al., 2014). Flow alteration can have negative consequences on the biotic composition, structure, and functioning of freshwater ecosystems (Bunn and Arthington, 2002). In addition to hydrological data, other relevant abiotic data in terms of water quality such as Dissolved Organic Carbon, pH, temperature etc. can provide an additional foundation for characterizing the presence of specific ecosystems. Historical data and/or modeling can be used to understand the reference conditions for the river hydrology whilst understanding ecosystem reference conditions requires historical data or similar conditions of another pristine river or branch (Pastor et al., 2014). Therefore, it is important to maintain and/or increase the monitoring of different flow states (flowing, and pools and drying) via the installation of gauges, via satellite images and the biological assessment of riparian vegetation, macrophytes, macroinvertebrates and aquatic fauna throughout the year. For example, some significant changes in communities groups and food web levels were found by Mor et al. (2018) who studied the gradient of macroinvertebrate assemblages (*Ephemeroptera*, *Trichoptera*) and found a difference in herbivorous and detritivorous species evolution along perennial and intermittent streams as a consequence of groundwater abstraction. A shift in habitats has also been identified in the Evrotas River in Greece as well as pollution consequences such as the creation of hypoxic conditions (Kalogianni et al., 2017). To help understand these changes tools exist to predict the hydrological and ecological consequences of hydrological alterations and associated management recommendations (Warfe et al., 2011).

6.3.3. Other flow alterations in IRES

Topography, soils, vegetation, and river/stream network topology are the main natural factors influencing water and sediment transport in temporary streams. Human activities may alter these catchment processes. Historically, intensive land use changes have transformed natural ecosystems into agricultural fields (Grove, 1996), markedly increasing soil erosion processes (Douglas, 1993). As deforestation results in increased sediment yields, the implementation of soil and water conservation practices can reduce negative impacts, thus resulting in reduced sediment yields (Estrany et al., 2010) even after perturbations (Garcia-Comendador et al., 2017). With the advent of rural mechanization and urban expansion in the 20th century, alterations in temporary streams suffered an exponential intensification. On one hand, massive abstractions, or other agricultural practices such as drainage or constructions in streams can exacerbate the drying up of temporary streams promoting an increase of channel bed storage of fine sediment and associated contaminants. On the other hand, urban expansion modifies hydrological processes, generating a continuous flow due to waste water (Estrany et al., 2011) or extreme flow and fine sediment transport responses as a result of combined sewer overflows (Old et al., 2003). Another impact on IRES is bedflow modification with, for example, the creation of impermeable bed surfaces leading to reduced infiltration rates into the bed during flood events (Goodrich et al., 2018).

6.4 Design and evaluation of Eflows adapted to IRES

6.4.1 Methodological frameworks for designing and evaluating Eflows

More than 200 Eflow methods have been developed so far worldwide. These methods are categorized into four types: hydrological, hydraulic, habitat-simulation and holistic methods (Pastor et al. 2014). While hydrological methods are easier and faster to apply than the others, they require a historical or simulated dataset of natural flow conditions. Habitat-simulation methods, on the other hand, are useful methods, especially when sensitive species have been monitored before any anthropogenic impacts. Holistic methods such as the Ecological Limits of Hydrologic Alteration (ELOHA) framework includes various components of the different types of methods with a concept linking hydrological alteration to ecological alterations. Although holistic methods are the most complete, they are also the most time and resource consuming and require frequent monitoring. Eflows started to be implemented in the US first with the Tessman method (1954) and then with the Tennant method (1976). Later, more holistic approaches were developed such in South Africa (Water Affairs and Forestry, 1997) and Australia. In Europe, there is not yet a uniform recommended Eflow method and this latter is linked with the Water Framework Directive that first requires the classification of ecological status of European rivers and Guidance Document No. 31 of the implementation of the WFD (European Commission, 2015). This latter gives guidelines for the assessment of Eflows with the aim to maintain or reach "good" ecological conditions. Eflows have barely been studied for IRES despite the latest studies recognizing their services (Datry et al. 2018).

6.4.2 Hydrological methods

- Eflow methods must be designed taking into account that the full range of variability of the flow regime is necessary to preserve river ecosystems (Poff et al. 1997; Arthington 2012).
- "Intermittency" is the key point in designing Eflows for IRES. Artificial permanency of the flowing phase should be avoided.
- Indicators of Hydrological Alterations (IHAs) that proved to be ecologically relevant for IRES have to be considered when setting an Eflow (Acuña et al. 2020; Richter et al. 1996).
- IHAs have to be evaluated over a prolonged period of time including dry and wet years (at least 20 years) to incorporate the high variability of streamflow that characterizes IRES (De Girolamo et al., 2017).
- IHAs describing duration and predictability of the dry phase (i.e. the number of months with non-flow and the six-month seasonal predictability of dry phase) must be considered (Gallart et al., 2012, D'Ambrosio et al. 2017).
- IHAs describing magnitude of monthly flow and annual minimum flow (i.e. min flow on 30-day and 90-day duration) must be considered when setting an Eflow to define the low flow condition, the frequency and timing of the transition phases (from flowing to connected pools, disconnected pools and dry riverbed) (Poff et al. 1997; Richter et al. 1996).
- IHAs describing the magnitude, duration, frequency, timing of high flows, as well as the rate of change, have to be considered for a comprehensive evaluation of Eflows.

- Methods for assessing the hydrologic alteration and the design of Eflows that explicitly take into account the occurrence of water pools after the cessation of flow should be progressively used in the practice, as the corresponding fundamentals and methods are already available.

A case study

To set an Eflow, an adaptation of the Range of Variability Approach to temporary waterways was defined and tested in the Celone River basin (SE, Italy; Acuña et al., 2020). After quantifying hydrological alterations, the following IHAs that represent specific ecological functions in IRES were estimated in natural conditions over 20-years (Table 6-2). The 25^{th} and 75^{th} percentiles were fixed as the minimum and maximum values for each IHA to design the Eflow regime. The method can be applied at different spatial scales, from small to broad-scale. It is a first level of analysis, simple, rapid, and inexpensive, suitable when data and time are limited as it requires only streamflow data (either measured or simulated). At the same time, this approach offers the advantage to define a hydrological regime consistent with the achievement of the environmental objectives of the WFD. Indeed, each selected IHA is suitable for enhancing a specific ecological function. The procedure is intended to be monitored and revised on biological data basis in a process of successive approximations able to identify the hydrological alterations compatible with the environmental objectives.

Table 6.2 Hydro-ecological functions

Flow component	Hydrological Indicators	Ecological function
Flow permanence; Predictability	Relative number of months with flow; Six-month seasonal predictability of dry period	To maintain structure of communities and preserve the development of taxa specialized in living in intermittent conditions.
Magnitude, duration and timing of annual extreme streamflow condition	Annual maxima of 1-day, 3-day, 7-day duration; Annual minima of 30-day, 90-day duration; Zero days; Date of minimum; Date of maximum; High pulse duration	To create sites for colonization, structure of river morphology and physical habitat conditions. To sustain the life cycle of native species and the richness of invertebrate assemblage. To guarantee the transition from connected to disconnected pool regime that accounts for communities variance.
Magnitude of monthly streamflow	Average monthly flow	To maintain species diversity and abundance and prevent the successful establishment of non-native species.
Frequency and rate of change	High pulse count; flashiness Index	To regulate community structure, guarantee population persistence. To prevent the establishment of non-native species, as in IRES native species are persistent to flash floods.

6.4.3 Methods for monitoring sediment

Information on the suspended sediment concentration in a stream will reflect the sampling method, sampling location and monitoring frequency, both in time and in stream cross-section (Gurnell, 1987). In addition, most fluvial sediment movement occurs during infrequent flood events (Evans et al., 1997). Therefore, infrequent routine sampling (e.g., weekly or monthly) may miss these events, leading to an underestimation of sediment transport (Gippel, 1995). Turbidity monitoring must be continuously undertaken to measure suspended concentration, considering additionally that extrapolation and interpolation procedures (Webb et al., 1997) and in areas with strong seasonal contrast cannot always be established (Sutherland and Bryan, 1989). The collection of temporal and spatial data for relevant hydrological processes, known as catchment monitoring, is crucial within the integrated catchment assessment framework. Catchment monitoring is complicated by the fact that the relevant processes operate at different temporal and spatial scales. This requires a nested approach to be adopted and that catchment data are collected across a micro to macro scale. The Na Borges River is a temporary stream located in the Mallorca Island where a nested approach was applied with four flow gauging stations equipped with continuous monitoring using data loggers linked to pressure and turbidity sensors. The nested approach demonstrated how sediment transfer is clearly determined by the seasonal alternation of influent and effluent discharges, and also human influences (Estrany et al., 2009).

6.4.4. Habitat-hydraulic models

Hydraulic-habitat models complement hydrological methods, by including in the analysis not only of flow discharge, but also hydromorphological features that are important for aquatic and riparian ecosystems. These features, such as water depth, flow velocity, substrate composition, presence of shelters and shore characteristics represent the physical habitat in which biotic communities live and develop. In IRES, during zero-flow periods, the occurrence of isolated pools and ponds and the connectivity between them, as well as the presence of thermal refugia due to shading of riparian vegetation, are also crucial for preserving local populations of aquatic biota. Thus, hydraulic-habitat models simulate the spatial and temporal variations of these physical habitat characteristics, which in turn, are used to predict species' presence and abundances (Ahmadi-Nedushan *et al.* 2006; Heggenes & Wollebaek 2013).

Common hydraulic-habitat models are based on the assumption that habitat changes according to river morphology and flow. However, in IRES when flow decreases to zero, the aquatic habitat is not reduced instantly but gradually. Despite the non-flow state, water can remain stagnant in pools and ponds for a few days or for a longer period of time providing habitat to water-related organisms. The amount of time pools are available as aquatic habitat depends on the morphology of the river stretch, the aquifer level, the soil humidity and the weather conditions (Acuña et al. 2020).

The mostly used hydraulic-habitat models working at the microscale (e.g., PHABSIM; Bovee 1982, and CASiMiR; Jorde et al. 2001) are based on hydrodynamic simulation models, which are suited for low gradient, perennial rivers, but are unreliable for flow rates near zero and challenging to apply in rivers characterized by flow intermittency (Acuña et al. 2020, Seaman *et al.* 2016). To assess aquatic habitat availability during both flow and non-flow conditions, Acuña et al. (2020) proposed an innovative approach able to evaluate and model aquatic habitat extent depending on flow and time passed after discharge interruption. Specifically, mesohabitat simulation models (MesoHABSIM, Parasiewicz et al., 2013, Vezza et al., 2014) integrated with a multi-scale hierarchical framework of stream geomorphic units (Belletti et al., 2017) were used to quantify habitat availability for fish, as an example of application of habitat modeling tools in temporary rivers.

Application of the MesoHABSIM method

An application of habitat-hydraulic models in temporary rivers was carried out in the Gaia river (Tarragona Province, Northeastern Spain; Clavero and Hermoso, 2015). The MesoHABSIM approach was used to model the spatio-temporal variation of physical habitat suitability for juvenile European eel (*Anguilla anguilla*) during both flow and non-flow phases. Juvenile European eel was selected as a target species, since it is an autochthonous and critically endangered fish species present in the Gaia river and could represent an important ecological target for the study. Multiple river surveys of each study site were performed to map hydromorphological units in a GIS environment. The interplay between flow, time and habitat availability was then elaborated by constructing a rating curve (called habitat-flow-time rating curve) to simulate the intermittent behavior of aquatic habitat availability. This curve is based on the habitat-flow rating curve of common habitat modeling tools to which is added, on the second quadrant of the graph, the recession of wetted area and habitat availability with respect to the amount of time after flow interruption (Acuña et al., 2020, Figure 6.1). Finally, the definition of this curve is used to generate the habitat time-series for temporary rivers that enables analysis of the spatio-temporal variation of habitat availability in the entire river segment.

Eflows design in temporary rivers should avoid increasing habitat bottle-necks by creating a continuous duration of minimum habitat availability. Meso-scale habitat models can be used to simulate possible scenarios of hydrological and morphological alterations of IRES, in order to select the most appropriate management strategy (see e.g., Koutrakis et al., 2018, Vassoney et al., 2019, Bussettini & Vezza, 2019).

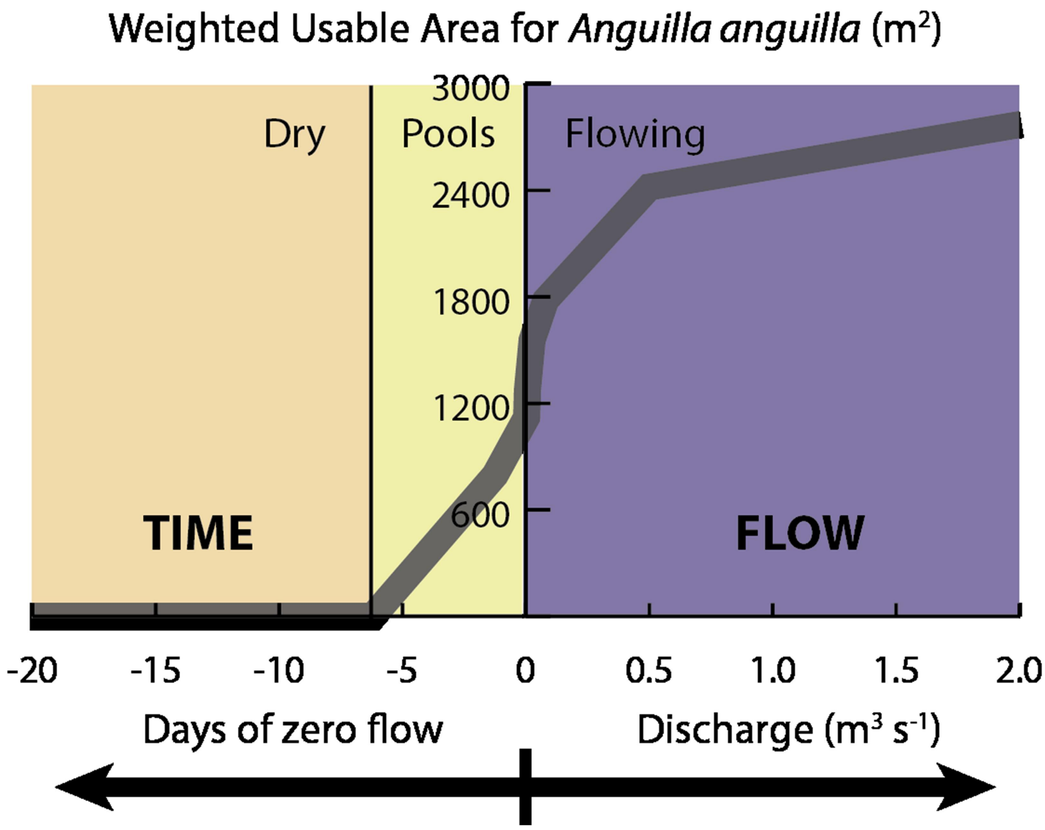

Figure 6.1 An example of habitat-flow-time rating curve for European eel, Anguilla anguilla (juvenile life stage). The curve refers to the application of the MesoHABSIM simulation model applied to the Gaiá river (Tarragona Province, Northeastern Spain). Modified from Acuña et al., 2020.

6.4.5 Economic valuation of Eflows

Economic valuation has been for long employed for accounting the social and economic benefits of implementing Eflows (Jorda-Capdevila and Rodríguez-Labajos, 2017a). They enable assessments to validate or reject a specific policy measure such as an Eflows restoration project, the purchase of irrigation rights, or the construction of a dam. But they can also be used to identify an optimal flow regime in terms of socio-economic benefits - see Chapter 5 which details different monetary and non-monetary methods to evaluate ecosystem services. They can be employed to assess either the whole policy measure (e.g. the contingent valuation method may ask a group of surveyees for their willingness to pay for the implementation a specific scenario of Eflows) or each of the ecosystem services provided by the studied IRES (e.g., the travel cost method for assessing the recreational value, the avoided cost method for the water purification).

In any case, for the design of Eflows, it is important to determine:

- ➔ What is the objective of an economic valuation in an Eflows assessment?
- ➔ What is the study area (e.g., a reach, a river, a catchment, or administrative area)?
- ➔ Who benefits from instream flows and who from water withdrawals?
- ➔ What ecosystem services are associated with free flowing river and what ecosystem services benefit from an alteration of the flow regime?

- ➔ How does the service provision change over time and space, and depends on the different components (e.g., duration, frequency, magnitude, timing) of different phases (e.g., hyperrheic, eurheic, arheic)?

From an economic valuation of Eflows, managers can obtain valuable information of societal preferences that can be included in methods to support decision-making, such as cost-benefit or multi-criteria analyses. Moreover, it provides basic information for more holistic methods of Eflows assessments.

6.4.6 Holistic methods

Holistic approaches are resource-consuming, step-wise structured frameworks for the collection, analysis and integration of data and knowledge to recommend flow regimes to meet specific objectives. In a co-construction process, they actively include stakeholders, adjustment of final conclusions and results through negotiation and consensus building. By including step-wise guidance on data and knowledge needs, holistic approaches are flexible and can be applied across a wide range of socio-ecological and biophysical conditions. They incorporate the hydrological and hydraulic-habitat methods described above, or expert knowledge in the absence of empirical data.

Interestingly, they allow integration of various biophysical variables, which cannot be integrated within hydrological and hydraulic-habitat methods due methodological and scale issues. These include, for example, the presence or absence of refuges during dry periods below the riverbed (e.g. the hyporheic zone), within the channel (e.g. the distance to the nearest perennial reach), or the landscape (e.g. the presence of parafluvial ponds) (Stubbington et al. 2017, Bogan et al. 2017). It also allows accounting for geomorphic variables that are ecologically relevant for organisms experiencing flow intermittence, such as the presence and life-span of pools in the riverbeds. However, currently the lack of flow-ecology relationships and their determinants in IRES is hampering the use of holistic approaches (e.g. Seaman et al. 2017). One of the most robust relationships quantified and tested across different climates and continents and organisms is flow intermittence (% of the year without flow) vs taxonomic richness (e.g. Datry et al. 2014, Leigh & Datry 2017). Thanks to SMIRES and the blooming field of IRES research, such types of transferable relationships should soon become available for the other biophysical variables mentioned.

6.5 Implementation process of Eflows in IRES

While progress has been made in the last decade in monitoring and designing new methods for IRES as shown in the previous sections, a challenge remains in implementing Eflows. In the section, we present the state-of-the-art of legal, technical and socioeconomic details of Eflow implementation procedure.

6.5.1 Legal considerations

There are large variations in Member State's legislation relating to Eflows at European level, which makes it difficult to set general legal considerations and suitable roadmaps to properly implement them. Information on environmental flow legislation in EU member States was recently compiled by the European Commission (European Commission, 2015). In cases where it is not possible to restore hydromorphological conditions and flow regime, the Water Framework Directive allows Member States to designate such water bodies as a “heavily modified water body” (HMWB), whose environmental objective is to reach “good ecological potential” instead of “good ecological status”. Most EU Member States have developed diverse provisions in their legislation referring to Eflow requirements and implementation in rivers to account for ecosystems needs, either at national or regional level. Thus, although we managed to calculate the Eflows needed to keep or achieve good ecological status, its implementation will require an additional and careful analysis to consider legal rights committed in water licenses, as well as social awareness, pressures by stakeholders for water use and economic costs. At the same time, environmental services gained by applying Eflows have to be taken into account in order to get a complete economic balance (Honey-Rosés, 2009). However, the implementation of Eflows poses significant challenges and can lead to a hard administrative process, especially for temporary streams. In some countries, water administrations have granted numerous water licenses that allow abstraction which exceeds the river flow, in most temporary rivers, especially during the dry season. Reduction or cessation of those licenses may take a long time, since they can give rise to numerous lawsuits. So, it is advisable to reach agreements with water users in order to find an efficient implementation procedure. However, water users will want to avoid losing production and negative economic effects. In some cases, it will be possible to meet technical or legal solutions to ensure the required environmental standards and minimize the economic impact. Therefore, it’s important to consider previous legal issues when implementing of Eflows:

- First, a thorough review of current uses and their efficiency is essential. A balance sheet and the current economic efficiency and social benefits for each water withdrawal and uses must be carefully analyzed and updated. Thus, inefficient uses and disproportionate water withdrawal can be considered in order to review reductions in water abstraction licenses. Most legislation allows review of inefficient or disproportionate uses to implement environmental requirements to preserve aquatic ecosystems. So, Eflows should be compatible with the more efficient water uses.
- On the other hand, water uses may have changed over the time, and the current status may not be completely covered by old water licenses. For instance, many water licenses for current hydropower uses were inherited from old water diversion licenses to power water wheels in old riverside factories. So, specific legal conditions for each water license should be thoroughly analyzed in order to account for current use, and included within Eflows standards.
- Another issue to consider is that the improvement of the natural environment acquires increasing value, as numerous activities are associated with the natural environment, while some currently consolidated uses may have lost performance. So, it should be noted that the preservation of a minimum environmental quality through the circulation of Eflows should be considered as new water services, even in temporary rivers.
- Lastly, the serious breaches of water license’s limits by water users could result in license cessation procedures, or amendment to new conditions which incorporate Eflows. Obviously, this procedure will depend on each nations water law. Nevertheless, this pathway could be explored when applying Eflows in water licenses or simply stopping them.

Legal status of Eflows in water laws and River Basin Management Plans (RBMPs) must be reinforced to properly implement them, and water authority regulators should have the ability to levy fines when Eflows are broken. The Eflow implementation has little chance of being implemented without suitable regulatory enforcement.

6.5.2 Technical considerations

Strategies for harmonizing water uses and Eflows need to be properly analyzed. Technical options, together with their economic implications, to meet agreement are suggested to avoid costly and time-consuming legal procedures (see above).

Here, we describe some strategies that can be considered to harmonize water use and Eflows. The strategy adopted for each water use will depend on their degree of compatibility with the Eflows regime, and regime of the available flow:

- Changes of water use over time. Some water users can maintain the same annual allocation and implement Eflow requirements by simply modifying the timing of when water is taken. Implementing this strategy could require a slight modification of their water license and, in some cases, structural changes on water withdrawal facilities. In order to minimize the economic impact, these costs could be partially subsidized. For example, in France, subsidies are given only for substitution, meaning that the volume stored cannot exceed the volume that was previously extracted in summer.
- Discontinuing water rights of expired licences. In some cases, water rights for some water uses expire soon (in the next 5 or 10 years). In these circumstances, the water license owner may be allowed to renew their water license, in exchange for the adoption of the Eflow measures.
- Negotiated agreements with holders of multiple licenses or a unique license that is shared between farmers. Users who manage multiple water withdrawals could be required to reduce some uses in exchange for increasing another. The aim would be to distribute water uses and maintain Eflows by looking for a comprehensive solution which allows reduced production loss.

6.5.3 Social considerations

Three main considerations should be taken into account when assessing Eflows (Jorda-Capdevila and Rodríguez Labajos, 2017). First, spatial and temporal heterogeneity plays a role in service provisioning but also in service demand. For example villagers in forested headwaters do not have anything to do with city dwellers or farmers in the lowlands. Second, there are many indirect beneficiaries that usually go unnoticed by the public administrations, or there are very different levels of influence between beneficiaries. For example shepherds that use the river channel as a path for the sheep do not participate in the public debates on water management. Third, clash of interests are more common than synergies in water management, so the recognition of trade-offs and a negotiation to preserve the ecosystem health, as well as the diversity of ecosystem services, is more suitable than looking for a win-win solution. Thus, the engagement of people is indispensable in order to promote social sustainability and guarantee a long-term implementation of the Eflows.

Public participation processes have been adopted throughout Europe to implement the WFD (Behagel & Turnhout, 2011), so, stakeholders contribute to discussions on strategies to harmonize existing water uses with the proposed Eflows, in addition to potential compensation schemes.

In order to implement a suitable Eflow policy, it is essential to undertake a public participation process that engages with stakeholders through a series of workshops, round-table discussions and meetings, during which water users can share their concerns about the Eflow regime implementation. To facilitate this process, Eflows need to be estimated first, and several implementation strategies proposed for them to be discussed (see above). Proposals have to be assessed in a participative process with owners, stakeholders, Non Government Organisations (NGO), environmentalists, fishermen, etc., providing critical information for water managers, and establishing a framework of consensus on Eflow implementation.

6.5.4 Economic considerations

Not all water bodies can reach good ecological status within an affordable cost. Some water bodies have a long history of severe modifications, motivated by the need to provide navigation services, flood protection, drought mitigation, hydropower production, or agriculture irrigation. However, a comprehensive evaluation, considering all costs and benefits, must be carried out before designating a water body as HMWB. Recent water management plans released by European water authorities showed a high percentage of HMWB (Tockner et al., 2012) and most European water management plans categorize between 10 and 30% of water bodies as HMWB, even reaching up to 60-70% in some industrialized areas. Nevertheless, most water bodies affected by water withdrawals and scarce flow regime can be restored within an affordable cost, when benefits of restoring environmental services are taken into account (Bardina et al., 2016).

The environmental costs associated with water withdrawals are not incurred by the owners of the hydroelectric facilities but rather passed on to third parties. Similarly, the restoration of Eflows in the river generate benefits that are distributed among the wider public (Lane-Miller et al., 2013). Environmental externalities are, by definition, inefficient from a societal perspective, since it would be preferable for water users to account for the environmental costs associated with their water use. Furthermore, depending on how much society values river ecosystems, it is possible that the total social costs incurred would be greater than the private gains acquired by the owners of the hydroelectric facilities. So, a comprehensive cost-benefit analysis has to be taken into account when restoring Eflows which also considers environmental goods and services.

6.6 Take-home messages and future directions

- In this chapter, we present an overview of the different types of Eflow methods that were developed since the Brisbane declaration in 2007 for both perennial and intermittent rivers. Some progress has been made in developing new classifications and indicators for IRES river types, incorporating the concept of aquatic phases, by determination of flowing water, pools and dry phases. Simulation habitat methods are also being developed with new eco-hydrological relationships for IRES. However, accessible data in terms of river discharge and associated sediment load is lacking for IRES.
- Increasing socioeconomic research is providing insight into ecosystem services provision and social values associated with IRES. This work highlights that implementing Eflows to IRES will improve not only the quality of riparian ecosystems, but also the livelihood of people that depends on them.

- There is a need to increase the number of flow and sediment gauging stations and the monitoring of ecological data (fish, macroinvertebrates and riparian vegetation), and also to promote public participation to achieve social and environmental sustainability goals.
- Since 2007, when the Brisbane declaration gave a definition to environmental flows, entire continents and states have been working on designing appropriate Eflow methods to be tested and implemented in their home-rivers. In Europe, the WFD requires the classification of ecological status of all European rivers, however, IRES were not always included in state member evaluation and poorly documented due to their short water flow durations and lack of valorization and monitoring of these latter. This is also the result of the criteria used by member states to delinate water bodies with for instance some countries that do not consider an IRES as a water body.
- The international scientific and water manager community including the SMIRES network have worked on different steps around the definition of Eflows, from the monitoring of hydrological and ecological parameters of IRES, to the definition of new tools and new methods that are now accessible to the community, until the socioeconomic challenges that water managers are facing during Eflow implementation. This work is constantly progressing and each chapter of this book will influence the definition and the design of adequate Eflow methods and propose new perspectives on how to implement Eflows in IRES.

7. Overview and Recommendations

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7.1 Introduction

Intermittent rivers and ephemeral streams (IRES) are watercourses that cease to flow and may dry, sometimes with connected or disconnected pools. Thus, they are characterized by alternating wet and dry phases, with or without pools on the streambed, and can support high biodiversity over the different phases, and endemic species. Intermittent streams typically cease to flow for several weeks or months each year, whereas ephemeral streams flow only for short periods, usually after rainfall events.

IRES are found throughout the world, where they provide a fundamental role within river systems and the wider environment. The ecological, socio-cultural and economic values of IRES are less well understood in comparison with perennial systems. IRES exhibit high variability in hydrological and morphological characteristics, acute changes and spatial and temporal differences in river physicochemistry and habitats, as well as the adaptation of biotic communities to shorter or longer dry phase periods. As a result, the management tools and methods applied to IRES may need to differ from those used in perennial rivers. For example, methods to assess the ecological status of rivers according to the Water Framework Directive were developed for perennial rivers, but their efficiency is not always proven for IRES and their protocols are not always suitable with the high variability of IRES.

In addition, ecosystem services provided by IRES differ between wet and dry phases and are affected by public perception in different socio-cultural contexts, which can cause conflict between stakeholder groups. The effects of pressures on IRES also differ from those in perennial rivers. IRES managers therefore have to deal with more and different challenges when they manage IRES. For example the ecological flows (Eflows) to restore IRES will require specific approaches that replicate natural dry conditions, and different implementation strategies. Managers may need to adapt their practices accordingly in order for their actions to achieve the desired objectives.

The purpose of this handbook is to help river managers to better understand the natural processes that affect and govern IRES ecosystems and their importance for biodiversity and local communities. This final chapter contains a synthesis of the current state of knowledge, based on the previous chapters of this handbook, and provides available tools to help water managers to better cope with IRES challenges.

7.2 Take-home messages

The “take-home” messages are summarized below according to the knowledge provided from the previous chapters of this handbook.

7.2.1 Hydrology and morphology of IRES

- Intermittence of flow in IRES is characterized by high variability, both in space and time, which may be natural or caused or changed by artificial influences such as water abstraction or releases. Intermittence may be natural or man-made. This must be established to ensure to select appropriate management targets and to apply ecological status assessment successfully.
- Temporal patterns in occurrence and connectivity of lotic, lentic, hyporheic and terrestrial habitats define the biologically relevant hydrological regime of IRES. Important aspects include the alternation of flowing, ponded and dry phases, and highly variable lateral, vertical, and longitudinal hydrological connections between stream surface waters and surrounding terrestrial and groundwater environments.
- Traditional hydrological approaches provide flow information from gauging stations such as periods of no flow and rates of change. However, these need to be complemented by direct observation or alternative techniques to characterize flowing, ponded and dry phases. Aquatic phase information can be obtained from various sources from modelling through to citizen science observations.
- Hydrological regime metrics and classifications are useful tools for describing habitats and hydrological alteration.
- Some hydrological regime metrics are useful to inform ecological status assessments in IRES. These include the presence of connected or disconnected pools, time between flow cessation and rewetting, zero flow period and seasonality of drying.
- Quantitative assessment of the relative influences of natural intermittence and human pressures remains a challenge that water managers need resolving to correctly manage IRES, preserve and restore them.

7.2.2 IRES are systems with dynamic physicochemistry.

- The spatial and temporal variability of water physicochemical characteristics is higher in IRES than in perennial rivers and streams.
- IRES hydrological regime has a strong influence on abiotic and biotic processes controlling stream physicochemical conditions. As a result, the magnitude and range of values for temperature, dissolved oxygen, pH, salinity, and concentrations of nitrate, phosphate and dissolved organic carbon are highly variable.
- Remarkable daily variation in physicochemical parameters is a common feature of IRES, especially as streams transition from the contraction to the dry phases. During this transition, spatial variation also increases.
- Biogeochemical processes occurring in dry channels have significant effects on water quality upon rewetting. These first-flush events have the highest

concentrations of nutrients and dissolved organic carbon, which may cause water quality issues downstream.

- Point source and diffuse pollution can have greater effects in IRES compared to perennial rivers, especially during periods of low surface discharge due to lack of dilution.
- Knowledge of the hydrological regime of IRES is pivotal for the proper design of water quality monitoring plans and for the correct interpretation of results from monitoring surveys. As an example, the selection of representative sampling locations depends on the aquatic phase in which sampling is undertaken.
- Restoration of impacted IRES to improve hydrological linkages and habitat connectivity with adjacent riparian zones and streambed sediments, can significantly enhance their nutrient removal capacity and mitigate effects associated with rewetting events or with human activities.
- Existing water physicochemical methods to assess ecological status can lead to misleading interpretation of quality when applied to IRES. To successfully monitor their ecological status, IRES-specific physicochemical quality standards must be established.
- Consideration of sediment characteristics and indicators of physicochemical function associated with dry and wet phases could enhance the assessment of the ecological status of IRES.

7.2.3 Community Ecology and Biomonitoring in IRES

- IRES instream habitats comprise flowing, ponded, hyporheic and dry patches that shift in space and time to support biodiverse communities of aquatic, semiaquatic and terrestrial species that have the potential to act as biomonitors of ecological status.
- IRES ecological quality is not assessed in many regions, and in regions where it is done, biomonitoring typically relies on methods developed for perennial systems. Current evidence shows that existing approaches do not work in IRES below a threshold of flow intermittence. Only where the performance of a standard biotic index has been shown to be effective in IRES should it be used to assess biological quality. Novel indicators/approaches are needed for reliable assessments of biological quality.
- Best practice involves the use of evaluated or specifically designed biotic indices that consider the aquatic communities present during IRES wet phases. In contrast, terrestrial communities remain unexplored as biomonitors of dry-phase quality but may be particularly useful in assessing ephemeral streams with short, unpredictable flowing phases.
- Characterization of communities indicative of unimpacted conditions in a particular IRES subtype is difficult, because spatial and temporal variability in community composition can impede description of a single reference condition.
- Future ecological quality assessments may be improved by functional metrics (which explore species traits, not their names), development of genetic tools, recognition of metacommunity dynamics, and by encompassing both aquatic and

terrestrial biotas. Managers should contribute to the testing and evaluation of these new approaches, to facilitate their application to ecosystem management.

- In extremely occasional or ephemeral streams, where bioindicators are not proven to be efficient, existing hydromorphological indicators can be more appropriate to assess ecological status. Alternatively, terrestrial biological fauna can be tested as well, together with other novel approaches.

7.2.4 Ecosystem services and social perception

- IRES provide a variety of benefits to our societies, by providing habitat for iconic species, regulating biogeochemical cycles, and providing raw materials such as water and timber, space for cultural manifestation and corridors for both wild and farmed animals.
- IRES can be of natural or from anthropogenic activity. This will influence the criteria and reference values used within assessments of ecosystem services provision and could ultimately determine a positive or negative overall evaluation.
- The value of IRES changes over time, varying with aquatic phase, season and over the years. Their value is not only intrinsic but depends on the sociocultural context, which may also change. Thus, by improving the condition, knowledge and awareness about IRES, managers can modify the way in which society perceives and values IRES.
- Some of the intrinsic and relational values associated with IRES are not well recognized, including sense of place, cultural identity, social cohesion and nature stewardship.
- There are many indicators that can be used to assess ecosystem service provision. Economic approaches involve monetization of the environmental and resource costs associated with IRES. Policy makers and managers need to consider these alternatives when developing an optimal approach to efficient water management.
- Non-monetary techniques consider people’s perception towards IRES and are helpful in incorporating instrumental, intrinsic and relational values into decision-making. They have the added benefit of acting as mutual learning projects and when inter-stakeholder conflicts occur.
- Public participation aids understanding of the multiple values of IRES and can improve social perception. Participatory mapping, citizen science, and scenario planning are some of the methodologies that can be employed.

7.2.5 Ecological/Environmental flows (Eflows)

- The terms “environmental flows” and “ecological flows” are both used and expressed as “Eflows”. The term “environmental flow” is a more comprehensive term that involves social and wider environmental issues. “Environmental flows” fulfil European and national legal requirements e.g. for protected areas. The term “ecological flow” is widely used by water managers when referring to the achievement of the environmental objectives according to the Water Framework Directive (WFD; 2000/60/EC), where the supporting hydromorphological elements, and suitable EFlows are used to achieve or maintain good or high ecological status in rivers.

- There are more than 200 methods for determining Eflows, but few of them are compatible with IRES. Eco-hydrological relationships must be studied beyond the flowing phase to include the pool and dry phases, to inform the design and implementation of suitable Eflows in IRES.
- Eflows should be implemented in light of ecosystem services and water rights according to each country's national law. Cost-benefit analysis to implement Eflows that enable achievement of good ecological status in IRES should also consider economic and social needs.
- Hydrological patterns in IRES, based on regime classification or hydrological metrics must be taken into account when setting Eflows.
- Habitat simulation methods are also being developed with new eco-hydrological relationships for IRES. However, accessible data describing river discharge and associated sediment load is lacking for IRES. New technology and modelling, complemented by citizen science schemes, will improve understanding the eco-hydrology of IRES and inform future Eflow approaches for IRES.
- Increasing socioeconomic research is providing insight into ecosystem services provision and social values associated with IRES, which aids the implementation of Eflows in IRES. So, implementing Eflows will improve not only the quality of IRES ecosystems, but also the livelihood of people that depends on them.

7.3 Future research needs

Future research is needed to address specific gaps in our understanding of IRES. Many uncertainties have to be resolved and challenges met in order to properly manage IRES in coming years. This section provides an overview of these research needs and how new knowledge could be applied to support the effective management of these systems. By working together, academics and managers can address these evidence gaps and improve how IRES are managed.

Additional and co-ordinated information on IRES needs to be collected:

A comprehensive effort towards the acquisition of cross-disciplinary information in IRES should be undertaken.

- Existing gauging stations in IRES with long data records should be maintained in order to improve our understanding of intermittence patterns (e.g. the duration and timing of the dry phase) and how these may alter as a result of climate change.
- Alternative techniques are needed to quantitatively detail and characteristic aquatic phases and how they change in space and time. These techniques may take advantage of modelling approaches complemented by direct observations when undertaking water or biological sampling or citizen science initiatives.

- The development of technological applications could be applied to enable the assessment of freshwater ecosystems and their conservation supported by monitoring. Remote-sensing could provide data on hydrology, stream morphology and physico-chemistry. These data could help to develop monitoring strategies. Tools should be developed to help managers to use these kind of data that requires at the moment some specific expertise.
- Information on human activities within the IRES catchments should be collected to help disentangling the causes of intermittence of flow (natural .vs. anthropogenic). Local information on cultural, social and economic context is also crucial to understand the different world views, ways of knowing, and public attitudes and perceptions that form the basis of values towards IRES. Incorporating these values will aid decision makers in addressing conflicts over IRES and enhance public perceptions and values regarding IRES.

We need to better understand IRES functioning:

- Understanding the respective influences of natural and anthropogenic causes of flow intermittence is a key priority.
- We need to understand whether ecosystems of natural and anthropogenically created IRES are similar and if they should be managed in the same way.
- We need to understand the three dimensional (lateral, vertical, and longitudinal) hydrological interactions in IRES and their respective influence on ecological and biogeochemical processes and how they may be affected by human activities (e.g. land-use changes, water abstraction or release, instream habitat restoration). This will help us understand how index performance varies in relation to intermittence, including identification of sampling time periods to best characterize ecological quality.
- A better understanding of sediment regimes, processes and the influence of standing vegetation and wood characteristics in IRES is also required.
- Further work is needed to understand biological response to intermittence, especially the effects on terrestrial species in dry channels. We need to test the generality/transferability of flow intermittence-ecology relationships.
- Evidence to help us disentangle the influences of natural intermittence and human pressures such as land-use change and channel modification are necessary to ensure effective management actions.

Better information on IRES characterization, chemical and nutrient cycle, and typology is required:

- Based on hydrological and morphological information, we need to develop an IRES typology across European water authorities, to recognize IRES specificities and provide the appropriate management tools and methods, for physicochemistry sampling or biomonitoring for example. This would provide the foundations to assess ecological status of IRES and more broadly for robust management of these dynamic ecosystems.

- Indicators of physicochemical function associated with dry and wet phases would also be relevant to assess IRES ecological status. The microbiota associated with sediments respond quickly to changes in hydrological and chemical conditions. Microbiota can thus reveal information about a stream's hydrological history as well as the dominant biogeochemical process associated with a given hydrological phase.
- Improving knowledge on nutrient and chemical dynamics is required to better restore IRES, especially in systems dominated with wastewater effluents.

How to classify ecological status in IRES:

- Where there are no suitable aquatic biological metrics, for example in ephemeral streams, assessments based on hydromorphology can indicate the naturalness of channel processes, thus enabling inference of ecological quality.
- Further evaluation of the performance of current biotic indices (most of which were developed to assess the ecological status of perennial streams) in IRES will identify which indices require adaptation or replacement. Indices need to characterize community responses to changing in-stream conditions (including ponding and drying).
- Identification of the reference condition of IRES remains a challenge. Characterizing biotic communities indicative of unimpacted conditions in different IRES subtypes and how they vary both spatially and temporally will be an important step.
- The boundaries between different status classes for the physicochemical quality elements, should be evaluated and if not appropriate should be redefined for IRES.
- The analysis of nutrient concentrations in sediments could be used as a surrogate for water quality assessment.
- Where there are no suitable biomonitoring approaches based on aquatic biota, for example in ephemeral streams, the development of new approaches is a priority. Multiple terrestrial communities, including invertebrates and plants show high potential to assess aspects of ecological status in dry channels.
- Novel indicators/approaches are needed for reliable assessments of ecological quality. These may include edna and molecular techniques or the incorporation of the ecological function of different biota.

We need a robust management of IRES:

- The broadening of the management of IRES towards socio-environmental disciplines is necessary to raise awareness of the local people and empower them to become actively involved in decision-making about their local environment.
- The combination of monetary and non-monetary techniques for the assessment of ecosystem services are necessary to understand multiple values associated with IRES. The integration of those values into decision making may help to reveal people's perceptions, to drive socially accepted management and to deal with inter-stakeholder conflicts. This integration is necessary especially to design and implement suitable Eflows.

7.4 Final Remarks

IRES are valuable and poorly explored river systems that need more research and monitoring to understand them and manage them effectively. This will only be achieved by academics, managers and stakeholders working together. The EU COST action CA15113 - the Science and Management of Intermittent Rivers & Ephemeral Streams (SMIRES) - brought together more than 350 hydrologists, biogeochemists, ecologists, modellers, environmental economists, social researchers and stakeholders from 33 different countries to develop a research network who synthesized the fragmented, recent knowledge on IRES. The results of this action, summarized within this handbook, have improved our understanding of IRES and translated this into a science-based guidance to aid sustainable management of river networks. However, much work is still needed to fully understand and appreciate these systems, which are expected to increase in extent as a result of climate change. We need to improve, develop and test tools for their effective management. The results of the project will be an important contribution to the implementation of the Water Framework Directive including the assessment of ecological status of IRES one of the subjects of the Common Implementation Strategy for the period 2019-2021. Besides that information provided in this handbook can help and guide future collaborative works to enhance the knowledge and needs to better manage IRES.

Case Studies

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Contributor authors *(alphabetic order): Monica Bardina, Anna Maria De Girolamo, Gerald Dörflinger, Sonia Fragoso, Giovanni Russo, Natashia Silva, Rania Tzoraki, Paolo Vezza, Benoît Terrier, Thibault Datry.*

The present chapter aims at illustrating the implementation of the different concepts and tools described in the previous parts of this Handbook, by presenting real-life case studies. These case studies comprise of projects implemented by local or national authorities and water managers in different countries. These include projects from Cyprus, Great Britain, Greece, France, Italy, Malta, Portugal and Spain. Each project had specific objectives according to the management issues experienced at each site and therefore relevant measures were put in place.

The illustrated case studies cover a range of restoration actions and measures to tackle management issues, including water abstraction, damming of rivers, degradation of river habitats, expansion of invasive alien species, protection of endangered species, floods etc. in the framework of IRES management. The actions refer to restoration of the riparian zone and stream habitats, establishment of ecological flows, morphological restoration and implementation of flood protection measures.

Each of the eight case studies presented, comprises of the following:

1. Description of the project and the relevant river basin,
2. General context of the case study and the problems to be solved,
3. Technical description of the project
4. Critical aspects/ Lessons learned / Recommendations for managers
5. Project monitoring
6. Relevant photo documentation and figures related to the case study.

Therefore, through these case studies, water managers can better understand the main issues arising while working on a restoration project in IRES and see in firsthand how these may be tackled along the way. Moreover, these paradigms may mirror management issues a water manager is facing in his district and implement identical or similar measures, knowing beforehand the key issues he will have to face and manage.

Yermasoyia river (Cyprus)
Cyprus River Basin District

Name of the Project: Hydromorphological restoration and restoration of the riparian zone in Yermasoyia river

Coordinates (WGS84): Upstream 507760m E 3844560m N
Downstream 508040m E 3839030m N

Start date: Oct 2015 **Date of the conclusion:** Dec 2015
Expected average lifespan: Lifetime
Cost: 55.000€ **Funding:** National funding
Responsible: **X** Public ☐ Private ☐ NGO

Objectives of the restoration project

1. Remove invasive alien plants from the river corridor and introduce suitable native species in order to support biodiversity.
2. Restore river continuity
3. Restore river banks' morphology and remove dumped materials.
4. Provide refugia for the critically endangered European Eel *Anguilla anguilla*
5. Increase habitats' diversity by creating suitable microhabitats

Hydrological characterization of the river basin

Climate: MAR (mm/year): 414 MAT (°C):25,4
Köppen class: Hot-summer Mediterranean climate (Csa)

Characteristics from the catchment at:
X the outlet ☐ the gauging station

Coordinates (WGS84): 508030m E 3839080m N
Catchment area (km²): 178,8
River length (km): 35,4
Elevation range (m a.s.l.): 0 to 1540

Geology: X Sedimentary ☐ Calcareous ☐ Siliceous ☐Unknown
X Other: Troodos ophiolite complex (upper and mid catchment)

Spatial pattern: The river mostly has an intermittent character with small spring-fed perennial stretches in parts of the catchment. Downstream Yermasoyia dam, the natural flow regime is harsh intermittent but flow releases from the dam for groundwater recharge lead to an extended flow period in a 4.5km long reach below the dam (but not up to the river mouth).

Seasonality: Intermittency in Yermasoyia basin has a typical Mediterranean character. Flows occur during the late autumn-spring period, followed by a dry phase from summer until mid-autumn. Downstream Yermasoyia dam, artificial flows at specific locations provide flows for longer periods. The lowest reach to the river mouth is ephemeral.

Main driver(s):
X Summer dry period ☐Freezing/snow **X** Water management
☐Interaction with groundwater ☐ Other:

Land use (forest and natural, agricultural, wetlands, artificial):
Forest: 68,5%; Agriculture: 24,5%; Urban: 3,6%;
Burned areas: 2,6%, Water bodies 0,7%

Main Pressures: **X** Dam upstream **X** Morphological alteration
X Water abstraction ☐ WWTP **X** Groundwater exploitation
☐ Livestock **X** Other: Agriculture

Absence of suitable refugia for the endangered European eel *Anguilla anguilla*.

Creation of deep ponds for the survival of *Anguilla anguilla*.

Context and issues to be solved

The restored section of Yermasoyia river is located downstream Yermasoyia dam (storage capacity 13,5Mm^3). It runs through suburban and urban areas and flows into Limassol coastal front. The dam construction some 5,5km upstream of the river mouth, resulted in a dramatic decrease of downstream flows and a subsequent alteration of river habitats, including dramatic changes in riparian vegetation. In addition, urban sprawling, dumping and trespassing, have degraded the riverine environment. Hydromorphological degradation was coupled with invasive species domination, such as the *Arundo donax, Acacia saligna* and *Eucalyptus gomphocephala,* which displaced local vegetation.

Due to the abstraction of natural flows, river flow occurs mainly due to controlled water releases from the dam through pipes, for groundwater recharge purposes. In addition, on rare occasions, dam overflow supplies the river with valuable water volumes. In these cases, the river corridor connects to the sea and eels have been recorded along the river stretch.

Technical description of the Project

Detailed planning of all restoration actions was undertaken and an environmental permission was granted. Restoration actions where undertaken at six distinct river stretches, covering a total area of 19.750 m^2. Restoration actions included:

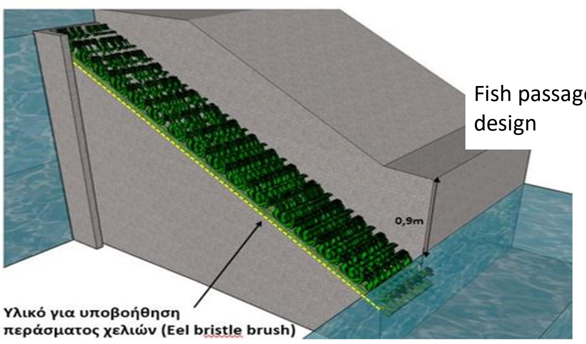
1. **Removal of alien invasive** species from the river banks and mainly the extensive stands of the giant reed *Arundo donax* by using only mechanical means such as heavy machinery (excavators) and manual labor work. During the vegetation removal, special care was taken to preserve all indigenous species found on the river corridor.
2. The removal of an old weir and the modification of a second instream structure by creating a fish pass, in order to **restore the longitudinal connectivity** of the river.
3. Creation of two deep pools in the riverbed, just upstream and downstream of the modified weir, which both progressively increase in depth reaching up to 2m and cover a total area of 850m^2. The pools where created to **enrich habitats diversity** and most importantly to **provide refugia for the endangered eel** and other aquatic species
4. **Creation of microhabitats** within the river channel by using logs and boulders and on the river banks by a creating variable bank morphology and enriching riparian vegetation.
5. **Plantation of 1725 plants of indigenous species** in various locations along the river banks, such as *Platanus orientalis, Nerium oleander, Vitex agnus-castus, Tamarix spp., Myrtus communis, Pistacia lentiscus, Ceratonia siliqua, Olea europaea,* etc. which are representative of the riparian galleries in Mediterranean intermittent rivers and the Mediterranean landscape. Plantlets survival was aided by the addition of fertile soil during planting and the installation of a watering system, as well as a maintenance programme.

Critical aspects/ Lessons learned / recommendations for managers

1. The eradication of the giant reed *Arundo donax* is a very difficult if not an impossible task, at least in Cyprus, and its suppression demands long-term dedicated efforts, a vast amount of money as well as hundreds of man-hours.
2. A maintenance programme is the most critical part for the overall success of a restoration project and should be implemented for at least 3-4 years after the plantings. Local authorities are the most suitable to implement maintenance programs, but their commitment must be ensured.
3. The boundaries of private land adjacent to the river, must be defined in situ by topographical survey work in order to avoid landowners' opposition. Additionally, in urban and suburban streams, underground pipes and cables may restrict the ability to implement restoration actions i.e. earthworks. Therefore, detailed investigations during the design stage are considered crucial for a subsequently successful restoration process.

Project monitoring

After the completion of the project, a maintenance programme must be put in place. Maintenance includes watering, weed removal and replacing dead plantlets. At the same time, the spread and development of the giant reed and other invasive species must be monitored and suppressed systematically by mechanical means.



Authors: Iakovos Tziortzis (itziortzis@wdd.moa.gov.cy), Gerald Dörflinger (gdorflinger@wdd.moa.gov.cy)
Water Development Department, Ministry of Agriculture, Rural Development and Environment, Cyprus.

River Clauge (FR)

Rhône River Basin District

Name of the Project: Morphological restoration of two temporary tributaries of the Upstream Clauge

Start date: 2007 **Date of the conclusion:** 2019

Expected average lifespan: not limited

Cost: around 600 000€

Funding: Various - European project Life, water agency, regional council, Universities

Responsible: X Public ☐ Private ☐ NGO

Objectives of the restoration project

1. To slow down the flows and restore similar hydrological conditions to those that existed in 1970s
2. Prevent regressive erosion and restore river and forest habitats
3. Raise the groundwater table in order to improve forestry production

☑ WFD ☑ Habitats Directive ☐ Flood Directive ☐ Other ________________

Hydrological characterization of the river basin

Climate:

MAR (mm/year): 1050 MAT (°C): 10.5°C

Characteristics from the catchment at:

☐ the outlet ☑ the gauging station: la Clauge à la Loye

Coordinates (°): lat. 5.540, long 47.03

Catchment area (km²): 116

Elevation range (m a.s.l.): 200 to 280

Geology: ☐Sedimentary ☐ calcareous ☐ siliceous ☐Unknown

☑ Other: old silicated alluvial deposits

Spatial pattern: Small tributaries of the river Clauge are naturally prone to intermittence. The forest infrastructures had exacerbated this intermittence and the river Clauge was drying at some point.

Seasonality: Lowest flows are experienced from June to September.

Main driver(s):

☑ Summer dry period ☐Freezing/snow ☑Water management ☑ Interaction with groundwater ☐ Other:

Land use (forest and natural, agricultural, wetlands, artificial):

Forest: 90%; Agriculture: 8%; Pastures -%; Urban: 2%; Other -%

Main Pressures: ☐ Dam upstream ☑ Morphological alteration ☐ Water abstraction ☐ WWTP ☐ Groundwater exploitation ☐ Livestock ☐ Other:

The straight bed of a tributary of the Clauge in 2005, prior to the remeandering works

A tributary of the Clauge in January 2009 after the remeandering works

Context and issues to be solved

For 70% of Clauge river length, it crosses the Forêt de Chaux, which is the second-largest continuous broadleaf forest in France (22,000 hectares). The original fish population consists of brown trout, chub, minnow, gudgeon, sculpin, monkfish, loach and brook lamprey that survives now only downstream of the project. The white-clawed crayfish was omnipresent in the past but disappeared. In 1950, foresters who were inspired by agronomists to believe that the presence of water tables in soils was harmful to trees, started draining, straightening and cleaning out around 80% of 400km of rivers in the forest of Chaux. Since the 1970s, the drying out of the river streams has been observed and the limits at which the flows are permanent have moved several hundred metres downstream. On the main river, a stretch of 7km which used to be permanent have become intermittent. The rapid removal of high water levels then caused major regressive erosion. The streams cut deeper channels and the river habitats became homogenised leading to reduced aquatic biodiversity. In 1973, 19 taxons of Ephemeroptera-Plecoptera-Trichoptera (EPT) had been sampled, compared to only 4 in 2006. Finally, the forestry operators in this area have observed a tendency towards dieback, especially in oak trees.

Technical description of the project

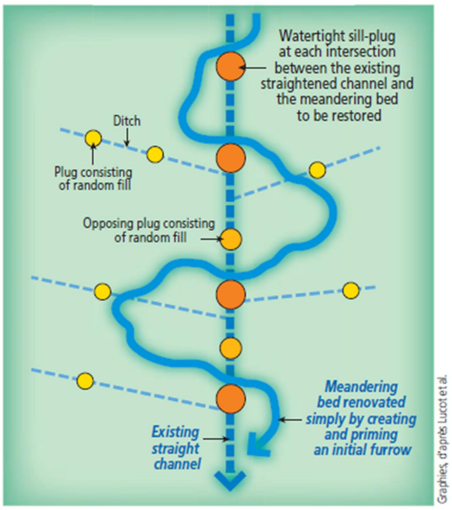
Following the success of the initial project in 2007, further river restoration was carried out between 2015 and 2019 on 40km of 32 tributaries and headwater streams. This stands for 45 km of streams as a total. The original meandering pattern was restored by eliminating the straight channel via a series of watertight “plugs”. These plugs are composed with wooden frames covered by a geotextile and material extracted on site. When streams were just cleaned out, or recalibrated without rectification, one-time sedimentary refill and fixed logjams were installed. In order to take better account of streams in forestry, a “water-oriented” access and exploitation scheme has been implemented by the ONF (French Forestry Commission). This scheme is designed to develop access routes, including river crossings, and to adapt the land division system according to the streams in order to reduce the impact of forestry practice on these streams. This scheme allows the number of culvert passages to be reduced by at least 30%.

Critical aspects/ Lessons learned / recommendations for managers

The monitoring over more than 10 years has revealed a modification of the hydric processes of the soils. The water table is shallower (-20 cm) and the variation in the groundwater levels is reduced. The remeandering allows water to remain in the soils for an additional fifteen to twenty days in springtime. The high water level has risen, while runoff is slower and therefore more favourable to aquatic fauna. A greater number of invertebrate taxons have been recorded after the works. The number of EPT has doubled between 2006 and 2018. A hitherto unseen species has been captured: the checkered caddisfly (included on the list of invertebrates threatened with extinction), which is a flagship species of wetlands. Long term monitoring (at least 10 years in this case) is necessary to assess the success of the stream restoration in its multiple dimensions (hydrological, biological etc.). Over time, the plugs become increasingly watertight and straight sections of the river are naturally filled in. The project requires minimal intervention for the foresters who are used to adopting a long-term approach. This operation also contributes to preventing the potential effects of global warming by taking immediate action in order to make soils cooler and increase water storage capacity. This experiment has been promoted by various articles and reports in the forestry sector. Information boards have also been erected. A technical day on the ecological restoration of streams in forested areas was also organized for stakeholders, state services, river managers. The final stage of this large scale project is the ecological restoration of the main river, the Clauge river, which should start in 2022.

Project monitoring

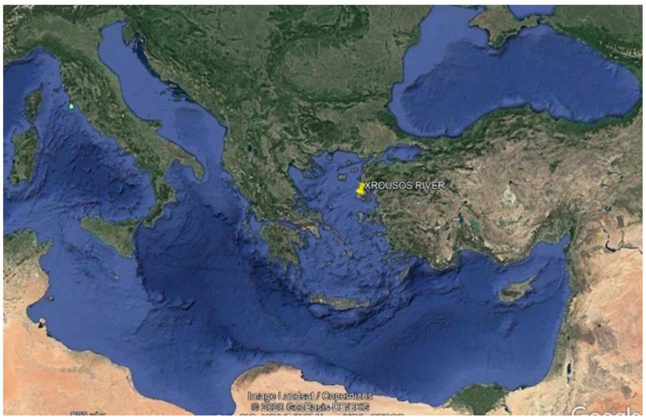
Pre-works monitoring was carried out by the Université de Franche-Comté in 2005-2006 and an initial monitoring operation was carried out from 2008. Soils, levels of the water table and aquatic invertebrates were studied. Piezometers were installed close to nine streams and within three of them. In parallel, benthos invertebrates were sampled in the Clauge and 6 tributaries as well as imagos were captured from the riverside vegetation using a butterfly-type net, each year. In addition, IRSTEA Lyon, a French research institute, conducted a 2 year-long survey of aquatic invertebrates along 8 sampling localities to explore how drying was determining these communities in space.



Author(s): Eric Lucot (UBFC), François Degiorgi (UBFC), Claire Magand (OFB), Thibault Datry (INRAE) and Benoît Terrier (AERMC). **Thanks to** Frédéric Sassard (ONF)

Xrousos (Chrousos) river, Lesvos island (Greece)

Aegean Islands River Basin District



Name of the Project: Xrousos Flood Protection Work
Coordinates WGS84 (°): Upstream 39,122907, 25,964882
Downstream 39,113017 25,964667
Start date: Dec 2019 **Date of the conclusion:** Mar 2020
Expected average lifespan: 40 yrs
Cost: 2.1 millions €
Funding: North Aegean Region, National funding
Responsible: **X** Public ☐ Private ☐ NGO

Objectives of the restoration project

1. Remove debris and sediment from the river corridor;
2. Restore river banks morphology (stabilizing river banks), increase the height of the rivers banks;
3. Prevent from floods

☐ WFD ☐ Habitats Directive **X** Flood Directive ☐ Other ___________________

Hydrological characterization of the river basin

Climate: Antissa Meteo Station
MAR (mm/year): 563 mm
MAT (°C): 17.6
Köppen class: Hot-summer Mediterranean climate (Csa)

Characteristics from the catchment at:
X the outlet ☐ the gauging station
Coordinates WGS84 (°): 39,106921 25,963519
Catchment area (km^2): 26.6
River length (km): N.A.
Elevation range (m a.s.l.): N.A.

Geology: ☐Sedimentary ☐ Calcareous **X** Siliceous ☐Unknown

Spatial pattern: The river has an intermittent character with small spring-fed perennial rivulets in parts of the catchment. The river delta forms the marvellous small wetland of Xrousos with code name Y411LES027, covering an area of 17,000 m^2. Part of the basin, near the estuary, belongs to the declared archaeological zone of Xrouso – Mesotopos.

Seasonality: Intermittency in Xrousos basin has a typical Mediterranean character. Flows occur during the late autumn-spring period, followed by a dry phase from summer until mid-autumn. The lowest reach to the river mouth is ephemeral.

Main driver(s): X Summer dry period ☐Freezing/snow **X** Water management **X** Intense Rain ☐ Other:

Land use (forest and natural, agricultural, wetlands, artificial): Forest: 20%; Agriculture: 15%; Agroforestry systems: 20%, Pastures 40%; Urban: 5%; Other __%

Main Pressures: ☐ Dam upstream **X** Morphological alteration **X** Water abstraction ☐ WWTP ☐ Groundwater exploitation ☐ Livestock ☐ Other:

Photo showing the river floodplain (Taken by facebook page "Ο χρούσος")

Photo showing the construction "Serazaneti" to stabilize the river bank and to enhance its capacity to accept higher stream power during the flood event (Taken by facebook page "Ο χρούσος")

Context and issues to be solved

The central branch of the Xrousos stream crosses the valley, meets the main bridge and outflows into the beach of Xrousos, west of Tavari village. It originates from the Ordimnos mountain range in central west Lesvos, northwest of the settlement Mesotopos and is divided into two large branches of which the western one is called Malonitas and the eastern Lagada. During flush floods (especially during the flood events of 29th Nov 2016 & 11th and 21th of Jan 2018) the water fills rapidly the main branch, breaks the existing protection cement constructions and overflow out of embankments resulting in flooding of well-cultivated areas and agricultural settlements. The section of the river that was restored is located close to the estuaries of the river. It runs through suburban and rural land, characterized by gravel and coarse sand bed material and low stability river banks.

Technical description of the Project

A detailed planning of all restoration actions was undertaken, but environmental permission was not asked due to the emergent character of the project. The restoration actions where undertaken at both banks of one river stretch along the waterbody covering a total length of 1955m. Restoration actions included a) the removal of debris and high volumes of the river bed material, b) construction of two sediment traps in the uphill part c) use of the river bed material as construction material for the "Serazaneti" blocks, d) the "Serazaneti" blocks wall length of 1955m in both river banks, for a height ranging between 2.5-4.5m.

Critical aspects/ Lessons learned / recommendations for managers

1. Future high flood events will test the capacity and strength of the project to mitigate the flood hazards. A maintenance programme should be implemented for many years after the flood protection works completion. Especially, following flood events, specific parts should be repaired and maintained.
2. During the construction, high mass volumes (sand, gravel and stones) were removed by the river bed and used for the "Serazaneti" blocks. The impact of high sand volumes not reaching anymore the adjacent Xrousos beach, an important touristic beach, was not considered.
3. Xrousos wetland is very small in size and often dry early enough to accommodate birds or support multiple pairs during breeding. Its high ecological value lies in the fact that it belongs to a network of small wetlands, which all together create a network of valuable bird feeders and refuges. However, not only Xrousos wetland but also all the small wetlands in the area need to be protected. The presence of significant species of amphibians and reptiles also necessitates the protection of the bare and dry land in the main part of the basin. The environmental impact of the flood protection project to the Xrousos wetland is not clear, since the project alters significantly the hydromorhological character and the hydrodynamic patterns.

Project monitoring

After the completion of the project, a maintenance programme must be put in place. Maintenance includes conservation of the "Serazaneti" blocks, removal of debris and sediment from the two sediment traps, monitoring of the ecological quality both the river and the wetland and monitoring of the hydromorphology of the sandy beach.

Author(s): Rania Tzoraki (rania.tzoraki@aegean.gr), Marine Science Department, University of the Aegean
Thanks to Stratis Boulboulis, Nicos Provatas, Lesvosnews for the photos and material.

Torrente Macinino (Italy)

Southern Apennines River Basin District

Name of the Project: Rinaturalizzazione con tecniche di ingegneria naturalistica delle sponde del Torrente Macinino
Coordinates (°): Upstream 41°54'38''N 16°08'17''E
Downstream 41°54'26''N 16°08'41''E
Start date: 18/03/2003 **Date of the conclusion:** 30/04/2004
Expected average lifespan: Lifetime
Cost: € 206.895,60
Funding: National Funding (Environmental Ministry)
Responsible: x Public ☐ Private ☐ NGO

Objectives of the restoration project

The main aim was to create a pilot project of "Good Practice" of restoration to be adopted in protected areas for recreating habitats for aquatic life, fauna and flora. At the same time, the bank of the eroded canal has been consolidated.

☐ WFD **X** Habitats Directive **X** Flood Directive ☐ Other ___________________

Hydrological characterization of the river basin

Climate:
MAR (mm/year): 536 mm
MAT (°C): 13°C (min. daily Temp); 20°C (max daily temp)
Köppen class: Cfa

Characteristics from the catchment at:
X the outlet ☐ the gauging station
Coordinates: 41°56'64" N 16°09'02" E.
Catchment area (km²): 32
River length (km): 36
Elevation range (m a.s.l.): 0 to 832

Geology: ☐Sedimentary **X** Calcareous ☐ Siliceous ☐ Unknown
☐Other

Spatial pattern: Torrente Macinino is a naturally Mediterranean intermittent river along its whole length.

Seasonality: Flow ceases along the river network during summer and only starts to flow after the first rain in autumn. Usually, it presents flowing conditions from November to May. During summer, the channel becomes fragmented into a series of isolated pools.

Main driver(s):
X Summer dry period ☐Freezing/snow ☐Water management
☐Interaction with groundwater ☐ Other:

Land use (forest and natural, agricultural, wetlands, artificial):
Forest: 60%; Agriculture: 25%; Agroforestry systems: 10%, Pastures 5%; Urban: 0%; Other 0%

Main Pressures: ☐ Dam upstream **X** Morphological alteration ☐Water abstraction ☐ WWTP ☐ Groundwater exploitation ☐ Livestock **X** Other: forest fires

Torrente Macinino. Before the project: Altered river morphology

Torrente Macinino. One year after the works of renaturation of the river morphology.

Context and issues to be solved

The Torrente Macinino flows into a forest catchment in the Gargano National Park classified as a SIC and ZPS Site due to its ecological integrity. In the mountain, the river network is ephemeral. In the middle course, some springs are present that are able to sustain a continuous flow for most of the year. In the lowland, especially in summer, connected and disconnected pools are present (area <0.50m^2 and deep <0.50m). Interconnection between surface and groundwater is limited to a few reaches. Summer is a critical period for aquatic life and for fauna. Alteration of the river morphology is the most important cause of habitat degradation having an impact on biogeochemical cycles, on river vegetation, and aquatic life. Riparian vegetation is degraded along several reaches, with a large extension of *Arundo donax* causing a decrease of diversity and number of refuge sites. Alien species, such as *Trachemys scripta* (*Tartaruga palustre americana*) and *Callinectes sapidus* (*Granchio blu*), that are competitors for food, space and oxygen, are devastating for juvenile and adult

Technical description of the Project

The restoration project covers the left bank (0,4km) of the river. Environmental engineering techniques such as “double log cribwall” (Palificata viva a due pareti) and “vegetated rock wall” (scogliera rinverdita) were used. On the one hand, the works are able to protect the enclosed road and, on the other hand, the restoration has an important role in the river ecology as the river morphology restoration was oriented to improve flora and fauna biodiversity. The cuttings used in the project were obtained through the trimming of *Tamarix africana* and *T. gallica,* which were present in the area of the site. The cuttings were integrated with *Phillyrea latifolia, Myrtus communis, Pistacia lentiscus, Teucrium fruticans, Viburnum tinus, Atriplex halimus, Spartium junceum, Laurus nobilis.*

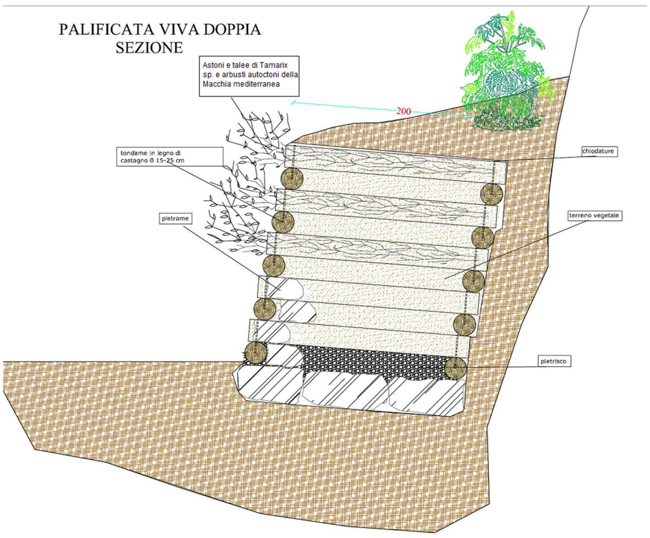
Lessons learned and recommendations to managers

The critical aspects were mainly due to the difficulties in realizing the works in continuum with the concrete banks realized in the past century. The project constitutes an important example of how the new techniques of restoration can be used for protecting aquatic ecosystems and biodiversity, improving water quality and providing cultural and recreational value to the site.

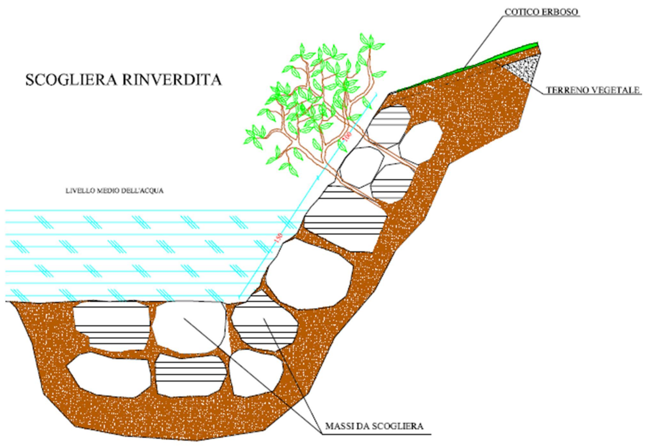
Project monitoring

The Project is monitored by the “Consorzio di Bonifica Montana del Gargano”. The results of the Project are very interesting and satisfactory. Along the river bank, after the restoration project, a riparian vegetation characterized by hydrophilic plants is present. Riparian zone has an important environmental, ecological and landscape value providing habitat biodiversity, soil conservation, and cultural and recreational value for citizens and tourists.

Project drawings



Double log cribwall



Vegetated rock wall

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Wied il-Fiddien (Malta)

Malta River Basin District

Name of the Project: Environmental Restoration of Wied Il-Qlejgħa
Coordinates (°): Upstream: 35°53'10.8"N 14°22'48.1"E,
Downstream: 35°53'32.9"N 14°23'27.5"E
Start date: May 2019 **Date of the conclusion:** December 2020
Expected average lifespan: lifetime
Cost: Studies: € 200,000; Works :€ 3,500,000
Funding: European Regional Development Fund
Responsible: **X** Public ☐ Private ☐ NGO

Objectives of the restoration project

1. To improve the structure of the target habitats
2. To reduce invasive, alien, and undesirable species;
3. To establish habitats that favour wildlife including avifauna;

X WFD ☐ Habitats Directive ☐ Flood Directive ☐ Other ____________________

Hydrological characterization of the river basin

Climate: MAR (mm/year): 553mm MAT (°C): 18.6^{0}C
Köppen class: Hot-summer Mediterranean climate(Csa)

Characteristics from the catchment at:
X the outlet ☐ the gauging station
Coordinates (°): 35°53'10.8"N 14°22'48.1"E
Catchment area (km²): 11.75
River length (km): 5
Elevation range (m a.s.l.): 239
Geology: ☐Sedimentary **X** Calcareous ☐ Ciliceous ☐Unknown ☐Other:

Spatial pattern: The water course within the catchments of the river basin predominantly incised in the Globigerina Limestone and Blue Clay formation, which account for the gently sloping and terraced valley sides. The valley floor has a mean slope of -0.072°. The valley bed is mainly characterised by soil / alluvial sediments and embankments / gabions. The uplands on both sides of the valley are karstified Upper Coralline Limestone plateaux. There are four major anthropogenic interventions: terracing, dams and impoundments, roads and bridges and modern embankments.

Seasonality: The Wied il-Fiddien river system is an intermittent stream that flows for between 8-10 months of the year depending on the rainfall pattern that year. Water starts flowing from early autumn, following the first heavy rains, and lasts throughout the winter and early spring season. Pools of standing water last into late spring and early summer.

Main driver(s):
X Summer dry period ☐Freezing/snow **X** Water management ☐Interaction with groundwater ☐ Other:

Land use (forest and natural, agricultural, wetlands, artificial):
Forest: 10%; Agriculture: 70%; Agroforestry systems: 0%, Pastures 0%; Urban: 5%; Other (roads) 15%

Main Pressures: ☐ Dam upstream ☐ Morphological alteration **X** Water abstraction ☐ WWTP **X** Groundwater exploitation ☐ Livestock ☐ Other:

The riparian habitat was previously dominated by a reedbed and other invasive species including *Ricinus communis* and *Arundo donax*. The watercourse was restricted to a central channel that was maintained clear through the passage of vehicles during the summer months.

The invasive species and reedbed were cleared, and replaced by a riparian woodland typical of the southern Mediterranean region, which was dominated by the rare *Salix pedicellata,* a Natura 2000 habitat listed in Annex I of the Habitats Directive.

Context and issues to be solved

Wied il-Fiddien in Malta, hosts one of the largest intermittent streams that flows for between 8-10 months per year, depending on the extent of precipitation. The stream passes through agricultural land through most of its watershed and is channelled and used as a source of water for irrigation. This creates pressure on the flow regime shortening the period of flow to less than 8 months during drier years. The high nutrient content of the water originating from agricultural land encourages the growth of nitrophilous species of the typical IRES habitats. In this project, various bioengineering techniques are used to restore the structure and functions of the riparian habitats found along the valleybed, including recontouring, replacement of invasive species by species typical of Willow and Poplar galleries of the central Mediterranean, and the slowing down of the water flow to increase aquifer replenishment.

Technical description of the Project

The ecological restoration focused on improving the structure and functions of the typical IRES habitats which were either highly degraded or absent along this stream. Works were also undertaken to stabilise the stream banks using various bioengineering techniques, slowing down flow to allow for the replenishment of groundwater. Moreover, the following restoration objectives were targeted:

1. to improve the structure of the restored habitats by restoring all the vegetation strata.
2. to reduce the negative impacts caused by invasive, alien and undesirable species.
3. to establish a spatial mosaic that maintains all trophic levels.
4. to minimise environmental pollution in the valley.
5. to establish a suitable substrate and self-sustaining habitats in the degraded areas that will need to be re-engineered through heavy works and the use of machinery.

These objectives will be achieved by focussing on developing eight habitat types. These habitats are based on target phytosociological associations that are typically found in valley and riparian ecosystems in the central Mediterranean.

Critical aspects/ Lessons learned / recommendations for managers

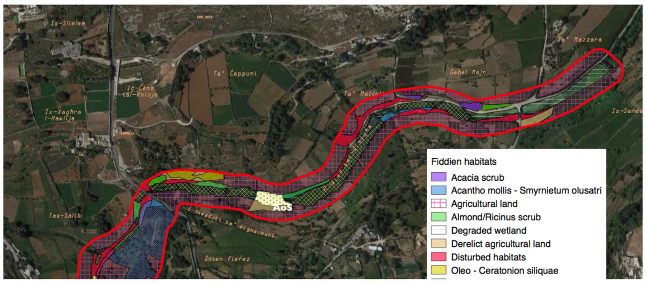
The three principles of ecological restoration strive to ensure that the restoration is effective, efficient and engaging. In this case, the project was effective as an in-depth baseline study was carried out that involved a landscape assessment, geohydrological, ecological, cultural and archaeological study and took these into consideration in the development of the restoration plan. The habitat restoration will build target reference ecosystems that will be the best examples of such habitats in the country, leading to the formation of the most extensive riparian ecosystems in Malta. The restoration project is also engaging as stakeholders have been involved at various stages of the process and had a central role in the decision making. Moreover, the site has been made accessible to these stakeholders through the setting up of a playground, a visitor centre and a walkway all along the site. The economic efficiency of the restoration was medium to poor as the project worked out to be expensive. The infrastructure costs of the project were high, and a cheaper design of the public area would have reduced the costs. Moreover, the project was implemented and managed by a contractor, rather than an NGO, raising the costs further.

Project monitoring

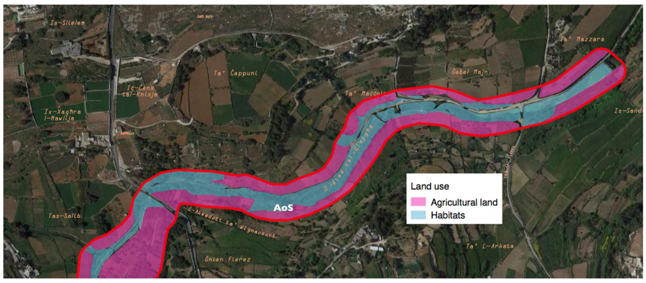
Number of individuals of *Procambrus clarkii* trapped per intervention effort. This data will provide information on the population size of this alien species. Monitoring is required to ensure the success of the proposed eradication method and to identify whether any additional measures were necessary.

Assessment of phytosociological association. The vegetation will be assessed according to the Braun-Blanquet system and attributed to the most appropriate Palaearctic habitat. This is done in the restored areas every five years.

Project drawings



Project drawings



Author(s): Eman Calleja, Institute of Applied Sciences, Malta College for Arts, Sciences and Technology, Fgura, Malta (eman.calleja@mcast.edu.mt)

Vascão River (Portugal)

Guadiana River Basin District Portugal

Name of the Project: LIFE Saramugo "Conservation of the Saramugo (*Anaecypris hispanica*) in the Guadiana basin (Portugal)" (LIFE13/NAT/PT/786; lifesaramugo.lpn.pt).

Coordinates (°): Upstream: lat 37.499081, long -7.705706, Downstream: 37.499139, -7.703872 (Datum WGS84)

Start date: July 2014 **Conclusion date:** December 2019

Expected average lifespan: Lifetime

Cost: 250.000,00 €

Funding: EU and National funding

Responsible: ☐Public ☐ Private **X** NGO

Objectives of the restoration project

To contribute to the long term conservation of Saramugo (*Anaecypris hispanica*), a freshwater endemic fish species of the south of the Iberian Peninsula, Critically Endangered, which only occurs in some tributaries of the Guadiana River and in a tributary of the Guadalquivir River. The project includes interventions in three rivers: Vascão, Murtigão and Safareja, although only the interventions in the Vascão River are here described:

- Rehabilitation of shelter, feeding and breeding habitats of the Saramugo, and of the connectivity between them;
- Improvement of habitat conditions for the species, in order to allow future reintroductions for population reinforcements with individuals bred ex-situ (safeguarding intra-specific diversity);

Besides that the project has also as objectives raising the: Involvement of landowners, farmers, fishermen and decision-makers; Public awareness about conservation issues of the endemic fish species, in particular the Saramugo, but also about conservation of riparian habitats and inhabiting species.

X WFD **X** Habitats Directive ☐ Flood Directive ☐ Other ___________________

Hydrological characterization of the river basin

Climate:
MAR (mm/year): 519, 2 MAT (°C): 18, 6
Köppen class: Hot-summer Mediterranean climate (Csa)

Characteristics from the catchment at:
☐ the outlet **X** the gauging station: Vascão (28L/02H)
Coordinates (°): 37.52; -7.579 (Datum WGS84)
Catchment area (km²): ~ 410 River length (km): ~ 200
Elevation range (m a.s.l.): 49 to 425

Geology: **X** Sedimentary ☐ Calcareous ☐ Siliceous ☐ Unknown
X Other: Metamorphic rocks (shale).

Spatial pattern: Vascão is a naturally Mediterranean intermittent river along its entire length.

Seasonality: Flow ceases in its whole length during summer and only starts to flow after the first rain in autumn. Usually, it presents flowing conditions from November to May. During summer, the channel becomes fragmented into a series of isolated pools.

Main driver(s):
X Summer dry period ☐ Freezing/snow ☐ Water management ☐ Interaction with groundwater ☐ Other.

Land use (forest and natural, agricultural, wetlands, artificial):
Forest/Agroforestry ~78%; %; Agriculture: ~20%: Urban: less than 2%.

Main Pressures:
☐ Dam upstream **X** Morphological alteration **X** Water abstraction ☐ WWTP ☐ Groundwater exploitation **X** Livestock **X** Other: Pollution; Expansion of exotic fish species;

Vascão River: absence of native riparian vegetation that would provide suitable habitat for the "*Critically endangered*" Saramugo fish.

Vascão River: rehabilitation of the natural morphological characteristics of river bank and of the native riparian vegetation.

Context and issues to be solved

The Vascão river flows into a very well preserved forest catchment integrated in the Guadiana Natural Park classified as a Ramsar Site due to its ecological integrity. It is also the largest river in Portugal without artificial barriers. Flow ceases in its whole length during summer and only starts to flow after the first rains in autumn, presenting flow conditions usually from November to May. During summer, the channel becomes fragmented into a series of isolated pools. Most of them are small and shallow (<50 m^2 in area and <50 cm deep) and persist in the section with bed rock substrate, having little or no contact with groundwater. Summer is the most critical time of year for the endemic freshwater fish and the pools are important summer refuges for those species, including the Saramugo. Most of the threats affecting Saramugo conservation result from the degradation of its habitats caused directly or indirectly by human actions such as water pollution, water abstraction and water retention by small rock weirs, and substrate removal. Other important pressures are degradation of riparian vegetation, with the expansion of *Arundo donax* (giant reed), which decreases the shade in the river and consequently the diversity and abundance of refuge sites; and the expansion of exotic fish species, such as *Micropterus salmoides*, *Lepomis gibbosus* and *Australoheros facetus* which compete for food, space and oxygen, with predation of fish eggs, juveniles and adults.

Technical description of the Project

The restoration actions taken by the LIFE Saramugo Project in Vascão River covered about 4. 736m^2 and have included: (1) Silt removal in riverbed, contributing to the maintenance of a large summer pool, to a more heterogeneous habitat mosaic with runs and riffles and to the improvement of water quality. Around 1 050m^3 of fine sediments were removed and used for the recovery of river banks morphology; (2) Removal of *Arundo donax*, both mechanically and manually, and subsequent restoration of river banks morphology using bioengineering techniques such as Salix spp. fences, vegetated crib walls and biorolls; (3) Rehabilitation of the riparian vegetation, in both river banks using riparian native species, characteristic from Mediterranean temporary rivers. The plants were obtained from seeds and branches of trees and scrubs from Vascão river banks, such as: *Nerium oleander, Tamarix africana, Myrtus communis, Rosa canina*, *Salix atrocinerea, Fraxinus angustifolia*. The survival of the plants was aided by the addition of fertile soil during planting. For each plant, a metal net protection was placed to protect it from herbivorous animals. A network for the protection of the Saramugo, "Os guardiões do Saramugo" was created, which includes regional authorities, municipalities, schools, private companies, farmers and citizens.

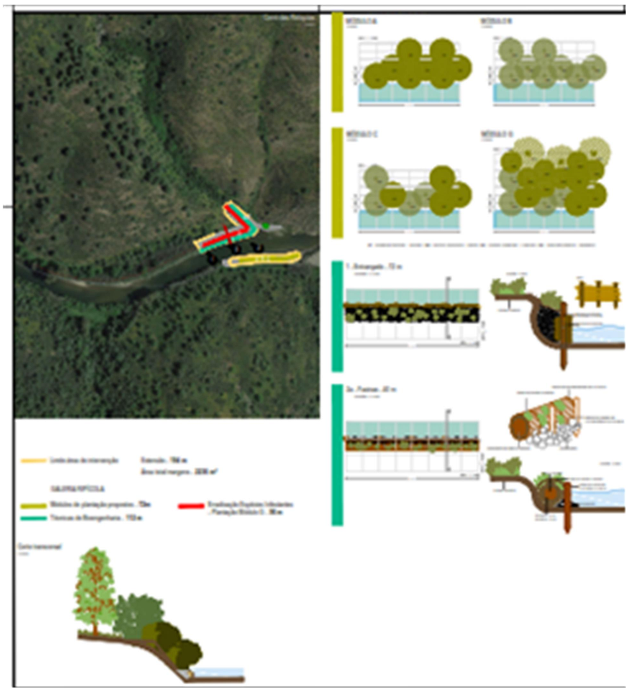
Critical aspects/ Lessons learned / recommendations for managers

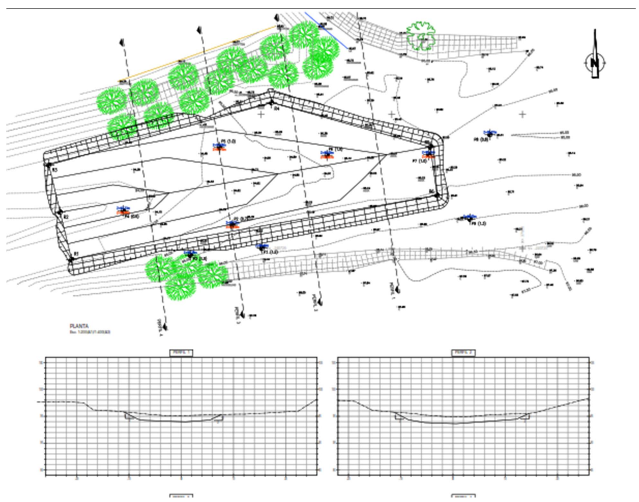
The main difficulties were: to obtain the authorisation of the landowners, sometimes requiring a long and complex negotiation process; the eradication of *Arundo donax*, very time consuming and expensive, requiring a lot of manual work; these interventions are expensive and require the use of heavy machinery, which limited the length of the river segments that were intervened; extreme climatic conditions, such as intense rain in winter or extreme heat during summer, affected significantly the success of the interventions and have caused delays; a maintenance programme is the most critical part for the overall success of a restoration project and should be implemented during the 3-4 years after the plantings.

Project monitoring

Some monitoring measures are foreseen, once the interventions carried out cannot be correctly assessed immediately after the conclusion of the project.

Project drawing river bank restore morphology and riparian gallery.



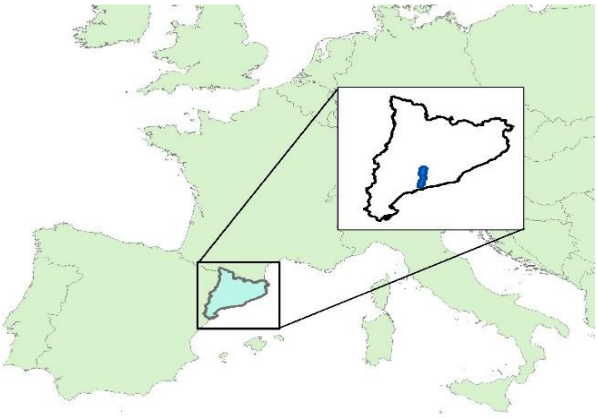


Author(s): Sónia Fragoso, Natasha Silva.
Liga para a Proteção da Natureza (LPN), Portugal (lpn.cea-castroverde@lpn.pt; www.lpn.pt)

Gaià River (Spain)

Catalan River Basin District (Catalonia)

Name of the Project: Restoring e-flow in the lower Gaia river
Coordinates (°): Upstream: 359869, 4561797 (UTM system)
Downstream: 362985, 4554623 (UTM system)
Start date: 2010 **Date of the conclusion:** in progress
Expected average lifespan: forever
Cost: Negligible. Action was carried out by reaching an agreement with dam's owner in order to enhance the reservoir management and yield. Reservoir is managed at low water levels in order to avoid water loss by infiltration (e-Flows can be released instead).



Objectives of the restoration project

Implementing e-flows in the lower Gaià River (NE Spain) affected by a big dam built for industrial water supply purposes. The water authority (the Catalan Water Agency - ACA), and Repsol Company (the owner of the Catllar dam) met a satisfactory agreement allowing a suitable e-flow without any significant unsustainable additional costs. E-flow was reached by enhancing the reservoir management and yield. Managing reservoir at low water levels allowed reducing water loss by infiltration. So, e-flows were released instead with a negligible cost and scarce impact on water availability for industrial purposes. A mesohabitat simulation model was applied to assess habitat availability for fish and e-flow requirements taking into account water intermittency. Additional flashing floods are also released to restore and maintain habitats.

Hydrological characterization of the river basin

Climate: MAR (mm/year): 450 MAT (°C): 17
Köppen class: Hot-summer Mediterranean (Csa)

Characteristics from the catchment at:
☐ the outlet X the gauging station: Vilabella
Coordinates (°): 360660, 4566070 (UTM system)
Catchment area (km²): 333.29
River length (km): 12 (from Catllar dam to the sea)
Elevation range (m a.s.l.): 143.1 to 0.

Geology: ☐Sedimentary X Calcareous ☐ Siliceous ☐Unknown ☐Other:

Spatial pattern: The Gaià river mostly has an intermittent character with small spring-fed perennial stretches in parts of the catchment. Downstream Catllar dam, the natural flow regime would be mainly intermittent with scarce flow or completely dry.

Seasonality: The Gaià river is a temporary river characterized by Mediterranean climate on the north-eastern Iberian Peninsula. Gaià has a total catchment area of 425 km^2, and a daily average discharge of 0.16 m3/s, which range from 0.04 to 0.26 m^3/s depending of dry or wet years

Main driver(s):
X Summer dry period ☐Freezing/snow X Water management
X Interaction with groundwater ☐ Other:

Land use (forest and natural, agricultural, wetlands, artificial):
Forest: 0%; Agriculture: 43%; Agroforestry systems: 54%, Pastures 0%; Urban: 3%; Other 0%

Main Pressures: X Dam upstream X Morphological alteration
X Water abstraction ☐ WWTP ☐ Groundwater exploitation
☐ Livestock ☐ Other:

A big dam was built in 1976 in the lower Gaià River, in order to store and supply fresh water for an oil refinery owned by Repsol. Water had not flowed from the dam after it was built, leaving 12 km downstream completely dry (from the dam to the sea).

An e-flow has been restored and tested for the last years, combining minimum in-stream flows together with controlled small released floods according to the natural flow regime upstream.

Ecological natural conditions

Catllar dam was built in 1976, following which the flow regime was completely removed from the lower Gaia river and river habitats lost together with their aquatic communities. Ancient local papers (previous to the dam building) document fishing activities carried out by citizens that lived alongside the lower Gaià river. Anguilla (*Anguilla anguilla*), a catadromous species that swims up the rivers after breeding in the sea was a common species together with *Barbus meridionalis*, etc.

Technical description of the Project

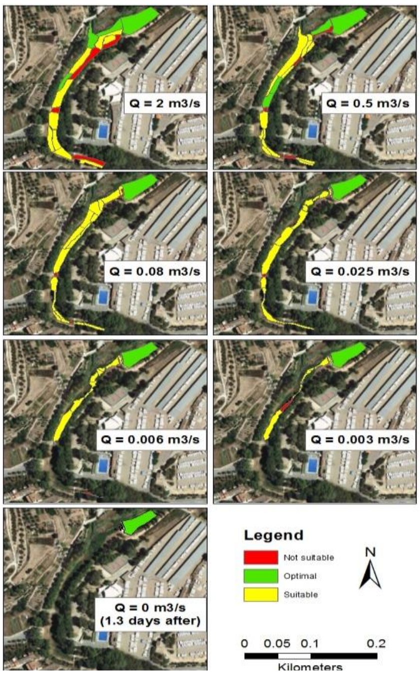
Environmental flows were calculated in the whole Catalan River Basin District by using hydrological methods. Results were later validated using fish habitat modelling through the Instream Flow Incremental Methodology (IFIM). The Government of Catalonia approved a Plan to restore Environmental Flows in 2006. In 2010 a satisfactory agreement was reached to restore minimum flows without significant water supply losses or additional costs by managing Catllar reservoir at low water level. A public commission composed by stakeholders, NGOs, local authorities and residents was set up in order to monitor changes in river and to analyse the agreement evolution. Four morphologically different river reaches, located downstream the Catllar dam, were selected and surveyed at the hydro-morphological-unit (mesohabitat) scale, collecting in each unit data on flow velocity, water depth, biotic and abiotic substratum type, cover and shelter availability for freshwater fishes (*A. anguilla* in its juvenile life stage).

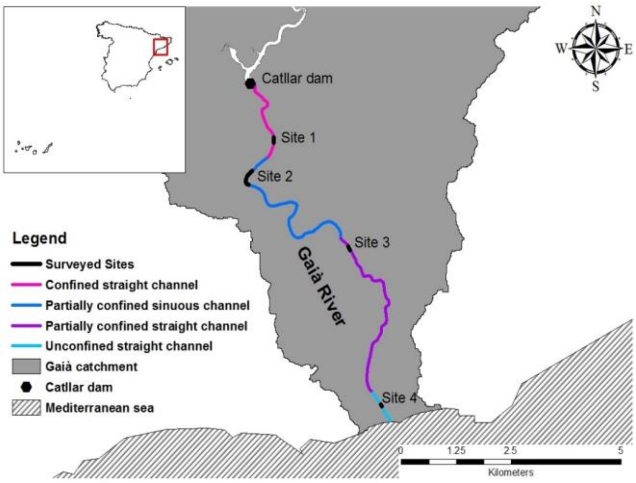
Critical aspects/ Lessons learned / recommendations for managers

Good experience of participation among the Water Authority, water users (industrial company and water irrigators), environmentalists (NGOs) and local citizens, who have improved water management. Water irrigators, initially opposed to the liberation of environmental flows, have verified that this new water management has not involved any loss of resource or consumption. Through a positive social impact, the population of the area has regained the contact with the river. The river has not been fully recovered yet, but the aquatic ecosystem has been partially improved and progressively recovered by biological communities. It shows the relevance of the minimum flows and small controlled floods to restore the river channel and aquatic habitats. Minimum flows are insufficient to connect river with the sea and to restore suitable habitats. It has been a positive and pioneering case of reservoir management adaptation that allows environmental flow without additional unsustainable cost or water warranty lost for local uses. It is also an example of water management based on the reservoir levels that can be useful in temporary rivers management.

Project monitoring

Over 20% of natural discharge has been released downstream of the Catllar dam in terms of e-flow during the last years. The river reach is partially recovered in the first 6 km out of 11 km from the dam to the sea. A total of 206 hydro-morphological units were described with flow discharge ranging from 0 to 2 m^3/s. Spatio-temporal variation of habitat availability was assessed by means of habitat-flow-time rating curves and habitat time-series were calculated. MesoHABSIM demonstrated flexibility and effectiveness in assessing habitat availability for fish in temporary rivers, being a suitable tool for further applications. The monitoring of physicochemical and biological indicators in the water body has been recovered.





Author(s): Antoni Munné (anmunne@gencat.cat); Mònica Bardina (mbardinam@gencat.cat); Paolo Vezza (paolo.vezza@polito.it)

River Misbourne (UK)

Thames River Basin District

Name of the Project: River Misbourne restoration
Coordinates (°): Upstream Lat. 51.710101, Long. -0.71264444
Downstream: Lat. 51.563573, Long. -0.48322403
Start date: 1990 **Date of the conclusion:** on-going.
Expected average lifespan: Not limited
Cost: On-going
Funding: Various, including water companies, Environment Agency, local interest groups.

Objectives of the restoration project

1. Reduce abstraction to restore a more natural flow regime and spatial and temporal intermittence
2. River restoration and barrier removal to address morphological degradation
3. Catchment management e.g. to create buffer zones and reduce fine sediment input.

X WFD ☐ Habitats Directive ☐ Flood Directive ☐ Other ___________________

Hydrological characterization of the river basin

Climate:
MAR (mm/year): 750 MAT (°C): 9.5
Köppen class: Temperate oceanic climate (Cfb)

Characteristics from the catchment at:
☐ the outlet **X** the gauging station: Denham Lodge
Coordinates (°): lat. 51.568562, long -0.49076307
Catchment area (km²): 94.8
River length (km): 27
Elevation range (m a.s.l.): 252,40 to 34,10

Geology: ☐Sedimentary X Calcareous ☐ siliceous ☐Unknown ☐Other:

Spatial pattern: Upper and mid-river dying with the extent and duration of drying dependent on groundwater levels.

Seasonality: Generally highest flows are experienced in January-March and lowest flows July-October following changes in groundwater levels. The extent and duration of drying varies depending on groundwater levels exhibiting seasonal and annual patterns.

Main driver(s):
☐ Summer dry period ☐Freezing/snow ☐Water management X Interaction with groundwater ☐ Other:

Land use (forest and natural, agricultural, wetlands, artificial):
Forest: 21%; Agriculture: 34%; Agroforestry systems: 0%, Pastures 28%; Urban: 17%; Other 0%

Main Pressures:

☐ Dam upstream X Morphological alteration X Water abstraction ☐ WWTP X Groundwater exploitation X Livestock ☐ Other:

River Misbourne Gerrards Cross golf course prior to morphological restoration June 2015

River Misbourne Gerrards Cross golf course after morphological restoration, June 2017.

Context and issues to be solved

The River Misbourne is a chalk stream. Chalk streams are a globally rare habitat, confined mainly to England and North West Europe, providing a home to a wide range of wildlife, including some of the UK's most threatened species.

Chalk streams are groundwater-fed and are characteristically shallow, clear and fast flowing. The intermittent 'winterbourne' reaches flow in response to changes in groundwater. Some of the rarest species living in chalk streams are especially adapted to living in winterbournes. In the 1980's the Misbourne was listed in the top 20 UK rivers most affected by abstraction.

The river channel is morphologically diverse, including sections that have been modified historically for land drainage, watercress cultivation, mill water provisioning, localised livestock poaching and more natural reaches.

https://environment.data.gov.uk/catchment-planning/WaterBody/GB106039029830

Technical description of the Project

In 1998 an alleviation of low flow (ALF) scheme was implemented, reducing groundwater abstraction. This has helped to increase flows and restore intermittence patterns in the upper sections. A second phase of work is now being considered to improve flow in the middle part of the river.

Alongside flow restoration measures are in place to address the effects of many centuries of human activity which have resulted in physical modifications. Morphological measures are undertaken to restore the natural bed and banks and aid the restoration of natural processes. Examples include a project to remove weirs and concrete lining of a section of the river flowing through a golf course. Previously all in channel vegetation was being removed to make golf ball retrieval easier and all the bankside vegetation was mown very close to the ground, offering limited river corridor habitat.

A combination of marginal shelves construction and woody material features were employed to establish more diversity in channel morphology. Selective management of vegetation of naturally established plants was carried out to encourage a diverse range of species, and to discourage plants from growing over the whole channel in low flows. *https://www.chilternsaonb.org/uploads/files/CCSP/misbourne_awareness_web.pdf*

http://www.colnecan.org.uk/index.php/the-action-plans/rivers-misbourne-and-alderbourne/rivers-misbourne-and-alderbourne-projects

Critical aspects/ Lessons learned / recommendations for managers

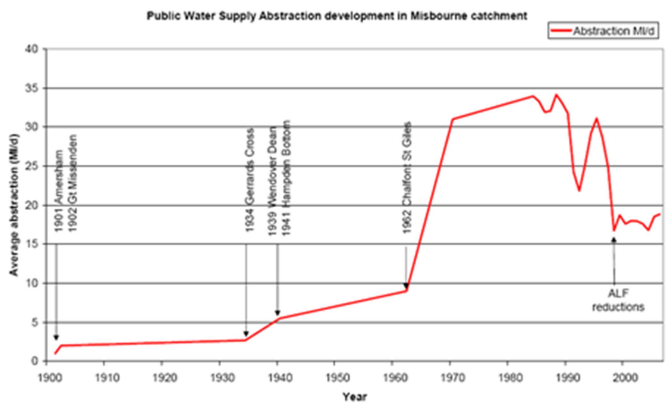
Addressing the multiple pressures affecting the Misbourne is complex, especially understanding the benefits of each individual measure applied so they can be prioritised. Understanding the baseline conditions, pressure and aspiration are essential to effective delivery. It is important to work with local communities and stakeholders to ensure they have a central role in the decision making process. Local interest groups are especially good at engaging with local land owners and promotes best practice in river side management.

Flow and morphological restoration should aim to restore natural processes including intermittence.

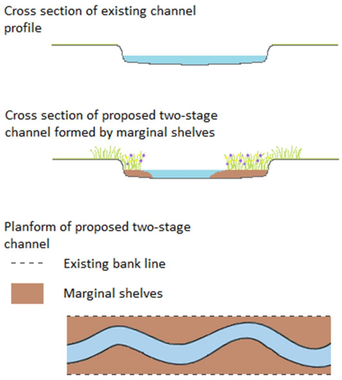
Project monitoring

Detailed monitoring of the ALF scheme was undertaken concentrating on recolonisation of formerly dry sections as flow and intermittence returned: Perrow, *et. al.* (2007): Life after low flow - ecological recovery of the River Misbourne. - British Wildlife. 18: 335-346. The study noted how drying promoted rare aquatic invertebrate species and richness as flows returned and highlighted the need for complementary measures to address physical degradation and insensitive river management. Monitoring is on-going including: flow states (flowing, ponded, dry), groundwater levels, citizen science river fly monitoring (http://www.riverflies.org/) and statutory agency WFD assessments.

Abstraction changes



Morphological restoration



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References

Acreman, M., Arthington, A. H., Colloff, M. J., Couch, C., Crossman, N. D., Dyer, F., ... & Young, W., 2014. Environmental flows for natural, hybrid, and novel riverine ecosystems in a changing world. *Frontiers in Ecology and the Environment*, *12*(8), 466-473. https://doi.org/10.1890/130134

Acuña, V., Datry, T., Marshall, J., Barceló, D., Dahm, C. N., Ginebreda, A., ... & Palmer, M. A., 2014. Why should we care about temporary waterways?. Science, 343(6175), 1080-1081. DOI: 10.1126/science.1246666

Acuña, V., Hunter, M., Ruhí, A., 2016. Managing temporary streams and rivers as unique rather than second-class ecosystems. Biol. Conserv. 211, 12–19. https://doi.org/10.1016/j.biocon.2016.12.025

Acuña V., Jorda-Capdevila D., Vezza P., De Girolamo A.M., McClain M.E., Stubbington R., Pastor A.V., Lamouroux N., von Schiller D., Munné A., Datry T., 2020. Accounting for flow intermittency in environmental flows design. *J Appl Ecol*. 2020; 57: 742– 753. https://doi.org/10.1111/1365-2664.13590

AFNOR, 2007. NF T90–354 – Water quality – Determination of the Biological Diatom Index (BDI). AFNOR (French Standards Association), Paris, France, 79 pp.

AFNOR, 2014a. NF EN 13946 – Water quality – Guidance for the routine sampling and preparation of benthic diatoms from rivers and lakes. AFNOR (French Standards Association), Paris, France, 18 pp.

AFNOR, 2014b. NF EN 14407 – Water quality – Guidance for the identification and enumeration of benthic diatom samples from rivers and lakes. AFNOR (French Standards Association), Paris, France, 13 pp.

Ahmadi-Nedushan, B., St-Hilaire, A., Bérubé, M., Robichaud, É., Thiémonge, N. & Bobée, B., 2006. A review of statistical methods for the evaluation of aquatic habitat suitability for instream flow assessment. River Research and Applications, 22, 503– 523. https://doi.org/10.1002/rra.918

Al-Qudah, O.M., Woocay A., Walton, J.C., 2015. Ephemeral stream chemistry below the elevation of near-zero net infiltration. Hydrological Process 29, 2385-2401. http://doi.org/10.1002/hyp.10375

Alba-Tercedor, J., Jáimez-Cuéllar, P., Álvarez, M., Avilés, J., Bonada i Caparrós, N., Casas, J., Mellado, A., Ortega, M., Pardo, I., Prat i Fornells, N., Rieradevall i Sant, M., 2002. Caracterización del estado ecológico de ríos mediterráneos ibéricos mediante el índice IBMWP (antes BMWP'). Limnetica, 21, 175–185. http://hdl.handle.net/2445/32903

Allen, D. C., Kopp, D. A., Costigan, K. H., Datry, T., Hugueny, B., Turner, D. S. & Flood, T. J., 2019. Citizen scientists document long-term streamflow declines in intermittent rivers of the desert southwest, USA. Freshwater Science, 38(2), 244-256. https://www.journals.uchicago.edu/doi/abs/10.1086/701483

Aparicio, E., Carmona-Catot, G., Moyle, P.B., García-Berthou, E., 2011. Development and evaluation of a fish-based index to assess biological integrity of Mediterranean streams. Aquat. Conserv.: Mar. Freshwater Ecosyst. 21, 324–337. https://doi.org/10.1002/aqc.1197

Armitage, P.D., Bass, J., 2013. Long-term resilience and short-term vulnerability of South Winterbourne macroinvertebrates. Proc. Dorset Nat. Hist. Archaeol. Soc. 134, 43–55.

Armitage, P.D., Moss, D., Wright, J.F., Furse, M.T., 1983. The performance of a new biological water quality score system based on macroinvertebrates over a wide range of unpolluted running-water sites. Water Res. 17, 333–347. https://doi.org/10.1016/0043-1354(83)90188-4

Armstrong, A., Stedman, R.C., Bishop, J.A., Sullivan, P.J., 2012. What's a stream without water? Disproportionality in headwater regions impacting water quality. Environ. Manag. 50, 849–860. https://doi.org/10.1007/s00267-012-9928-0

Arthington, A., 2012. Environmental Flows. Saving Rivers in the Third Millennium. 455 University of California Press, Berkeley, California.

Assendelft, R. S., & van Meerveld, H. J., 2019. A Low-Cost, Multi-Sensor System to Monitor Temporary Stream Dynamics in Mountainous Headwater Catchments. Sensors, 19(21), 4645.

Azarnivand, A, Camporese, M, Alaghmand, S, Daly, E., 2020. Simulated response of an intermittent stream to rainfall frequency patterns. Hydrological Processes. 2020; 34: 615–632. https://doi.org/10.1002/hyp.13610

Bagstad, K.J., Stromberg, J.C., Lite, S.J., 2005. Response of herbaceous riparian plants to rain and flooding on the San Pedro River, Arizona, USA. Wetlands 25, 210–223. https://doi.org/10.1672/0277-5212(2005)025[0210:ROHRPT]2.0.CO;2

Baldwin, D.S., Mitchell, A.M., 2000. The effects of drying and re-flooding on the sediment and soil nutrient dynamics of lowland river-floodplain systems: a synthesis. Regul. Rivers: Res. Manage. 16, 457–467. https://doi.org/10.1002/1099-1646(200009/10)16:5%3C457::AID-RRR597%3E3.0.CO;2-B

Baldwin, D.S., Wallace, T.A., 2009. Biogeochemistry. In Ecological Outcomes of Flow Regimes in the Murray-Darling Basin, eds. I.C. Overton, M.J. Colloff, T.M. Doody, B. Henderson, S.M. pp. 47–57. Canberra: CSIRO.

Beaufort, A., Lamouroux, N., Pella, H., Datry, T., Sauquet, E., 2018. Extrapolating regional probability of drying of headwater streams using discrete observations and gauging networks. Hydrology and Earth System Sciences, 22(5), 3033-3051. https://doi.org/10.5194/hess-22-3033-2018

Beaufort, A, Carreau, J, Sauquet, E., 2019. A classification approach to reconstruct local daily drying dynamics at headwater streams. Hydrological Processes, 33: 1896– 1912. https://doi.org/10.1002/hyp.13445.

Beentjes, K.K., Speksnijder, A.G., Schilthuizen, M., Schaub, B.E., van der Hoorn, B.B., 2018. The influence of macroinvertebrate abundance on the assessment of freshwater quality in The Netherlands. Metabarcoding Metagenomics 2, e26744. https://doi.org/10.3897/mbmg.2.26744

Belletti, B., Rinaldi, M., Bussettini, M., Comiti, F., Gurnell, A.M., Mao, L., Nardi, L. & Vezza, P. (2017) Characterising physical habitats and fluvial hydromorphology: A new system for the survey and classification of river geomorphic units. Geomorphology, 283, 143–157. https://doi.org/10.1016/j.geomorph.2017.01.032

Bhamjee, R., & Lindsay, J. B., 2011. Ephemeral stream sensor design using state loggers. Copernicus Publications.

Birk, S., Bonne, W., Borja, A., Brucet, S., Courrat, A., Poikane, S., Solimini, A., van de Bund, W., Zampoukas, N., Hering, D., 2012. Three hundred ways to assess Europe's surface waters: an almost complete overview of biological methods to implement the Water Framework Directive. Ecol. Indic. 18, 31-41. http://dx.doi.org/10.1016/j.ecolind.2011.10.009

Bogan, M.T., Boersma, K.S., 2012. Aerial dispersal of aquatic invertebrates along and away from arid-land streams. Freshwater Sci. 31, 1131–1144. https://doi.org/10.1899/12-066.1

Bogan, M.T., Lytle, D.A., 2007. Seasonal flow variation allows 'time-sharing' by disparate aquatic insect communities in montane desert streams. Freshwater Biol. 52, 290–304. https://doi.org/10.1111/j.1365-2427.2006.01691.x

Bonada, N., Dolédec, S., Statzner, B., 2007. Taxonomic and biological trait differences of stream macroinvertebrate communities between mediterranean and temperate regions: implications for future climatic scenarios. Glob. Chang. Biol. 13, 1658–1671. https://doi.org/10.1111/j.1365-2486.2007.01375.x

Bonada, N., Rieradevall, M., Prat, N., Resh, V.H., 2006. Benthic macroinvertebrate assemblages and macrohabitat connectivity in Mediterranean-climate streams of northern California. J. N. Am. Benthol. Soc. 25, 32–43. https://doi.org/10.1899/0887-3593(2006)25[32:BMAAMC]2.0.CO;2

Bou, C., Rouch, R., 1967. Un nouveau champ de recherches sur la faune aquatique souterraine. C. R. Acad. Sci. de Paris. 265, 369–370.

Boulton, A.J., Valett, H.M., Fisher, S.G., 1992 Spatial distribution and taxonomic composition of the hyporheos of several Sonoran Desert streams. Arch. Hydrobiol. 125, 37–61.

Boulton, A.J., Stanley, E.H., 1995. Hyporheic processes during flooding and drying in a Sonoran Desert stream. II. Faunal dynamics. Arch. Hydrobiol. 134, 27–52.

Boulton, A., Brock, A.M., 1999. Australian Freshwater Ecology: Processes and Management. Gleneagles Publishing.

Boulton, A.J., 2000. The subsurface macrofauna. In: Jones, J.B., Mullholland, P.J. (Eds.) Streams and Groundwaters. London, Academic Press, pp. 337–361. https://doi.org/10.1016/B978-012389845-6/50015-6

Boulton, A.J., 2003: Parallels and contrasts in the effects of drought on stream macroinvertebrate assemblages. – Freshwater Biol. 48: 1173–1185. https://doi.org/10.1046/j.1365–2427.2003.01084.x

Boulton A.J., Lake P.S., 2008. Effects of drought on stream insects and its ecological consequences. In: Lancaster J., Briers R.A. (Eds.), Aquatic Insects: Challenges to Populations. CAB International, Wallingford, UK, pp. 81–102. http://dx.doi.org/10.1079/9781845933968.0081

Boulton, A.J., Datry, T., Kasahara, T., Mutz, M., Stanford, J.A., 2010. Ecology and management of the hyporheic zone: stream-groundwater interactions of running waters and their floodplains. J. N. Am. Benthol. Soc. 29, 26–40. https://doi.org/10.1899/08-017.1

Boulton, A.J., 2014. Conservation of ephemeral streams and their ecosystem services: what are we missing? Aquat. Conserv. Mar. Freshwat. Ecosyst. 24, 733–738. https://doi.org/10.1002/aqc.2537

Boulton, A. J., Rolls, R. J., Jaeger, K. L., & Datry, T., 2017. Hydrological connectivity in intermittent rivers and ephemeral streams. In Intermittent rivers and ephemeral streams (pp. 79-108). Academic Press.

Bourke, S. A., Shanafield, M., Hedley, P., and Dogramaci, S. (in review, 2020) A hydrological framework for persistent river pools in semi-arid environments, Hydrol. Earth Syst. Sci. Discuss., https://doi.org/10.5194/hess-2020-133.

Bovee K.D. (1982) A Guide to Stream Habitat Analysis Using the Instream Flow Incremental Methodology. U.S. Fish and Wildlife Service, Fort Collins, Colorado, U.S.A.

Brisbane Declaration, (2007, September). The Brisbane Declaration: environmental flows are essential for freshwater ecosystem health and human well-being. In *10th International River Symposium, Brisbane, Australia* (pp. 3-6).

Brown, L.E.. Hannah, D.M., 2008. Spatial heterogeneity of water temperature across an alpine river basin. HydrolProcess 22, 954-967. http://doi.org/10.1002/hyp.6982

Brummer, M., Rodríguez-Labajos, B., Nguyen, T. T., & Jorda-Capdevila, D. 2017. "They have kidnapped our river": Dam removal conflicts in Catalonia and their relation to ecosystem services perceptions. Water Alternatives 10 (2017), Nr. 3, 10(3), 744-768.

Bruno, D., Belmar, O., Sánchez-Fernández, D., Velasco, J., 2014. Environmental determinants of woody and herbaceous riparian vegetation patterns in a semi-arid mediterranean basin. Hydrobiologia 730, 45–57. https://doi.org/10.1007/s10750-014-1822-8

Bruno, D., Gutiérrez-Cánovas, C., Sánchez-Fernández, D., Velasco, J., Nilsson, C., 2016b. Impacts of environmental filters on functional redundancy in riparian vegetation. J. Appl. Ecol. 53, 846–855. https://doi.org/10.1111/1365-2664.12619

Bruno, D., Gutiérrez-Cánovas, C., Velasco, J., Sánchez-Fernández, D., 2016a. Functional redundancy as a tool for bioassessment: A test using riparian vegetation. Sci. Total Environ. 566, 1268–1276. https://doi.org/10.1016/j.scitotenv.2016.05.186

Buffagni, A., Stocchetti, E., Armanini, D.G., Demartini, D., 2012. Establishment of an Assessment Method for the Biological Quality Element 'Benthic Invertebrates' in Cyprus Temporary Rivers and Participation in the Corresponding Intercalibration Exercise 2010-2011 for the Implementation of the Water Framework Directive 2000/60/EC. Tender Procedure No.: TAY26/2010. Ministry of Agriculture, Natural Resources and Environment – Water Development Department, Republic of Cyprus, p. 117.

Bunn, S.E., Arthington, A.H., 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environ. Manage. 30, 492–506. https://doi.org/doi:0.1007/s00267-002-2737-0

Burkhard, B., F. Kroll, S. Nedkov, and F. Müller. 2012. Mapping ecosystem service supply, demand and budgets. Ecological Indicators 21:17–29. https://doi.org/10.1016/j.ecolind.2011.06.019

Bussettini, M. and Vezza P., 2019. Guidance on Environmental Flows, Integrating Eflow Science with Fluvial Geomorphology to Maintain Ecosystem Services. World Meteorological Organization - Commission for Hydrology. WMO 1235, pp. 52.

Cavalli M., Trevisani S., Comiti F., Marchi L., 2013. Geomorphometric assessment of spatial sediment connectivity in small alpine catchments. Geomorphology, 188, 31-41. doi: 10.1016/j.geomorph.2012.05.007

Callow, J. N., & Boggs, G. S., 2013. Studying reach-scale spatial hydrology in ungauged catchments. Journal of hydrology, 496, 31-46. https://doi.org/10.1016/j.jhydrol.2013.05.030

Cañedo-Argüelles, M., Bogan, M.T., Lytle, D.A., Prat, N., 2016. Are Chironomidae (Diptera) good indicators of water scarcity? Dryland streams as a case study. Ecol. Indic. 71, 155–162. http://dx.doi.org/10.1016/j.ecolind.2016.07.002

Castro, A.J., García-Llorente, M., Martín-López, B., Palomo, I., Iniesta-Arandia, I., 2013. In: Di Bella, C., Alcaraz-Segura, D. (Eds.), Multidimensional Approaches in Ecosystem Services Assessment. Earth Observation of Ecosystem Services. Taylor & Francis, Boca

Castro, A.J., López-Rodríguez, M.D., Giagnocavo, C., Gimenez, M.,Céspedes, L.; La Calle, A., Gallardo, M., Pumares, P., Cabello, J.; Rodríguez, E., Uclés, D., Parra, S., Casas, J., Rodríguez, F., Fernandez-Prados, J.S., Alba-Patiño, D., Expósito-Granados, M., Murillo-

López, B.E., Vasquez, L.M., Valera, D.L., 2019. Six Collective Challenges for Sustainability of Almería Greenhouse Horticulture. International Journal of Environmental Research and Public Health, 16, 4097. https://doi.org/10.3390/ijerph16214097

Chadd, R.P., England, J.A., Constable, D., Dunbar, M.J., Murray-Bligh, J.A., Wood, P.J., Extence, C.A. Leeming, D.J., 2017. An index to track the ecological effects of drought development and recovery on riverine invertebrate communities. Ecol. Indic. 82, 344–356. http://dx.doi.org/10.1016 /j.ecolind.2017.06.058

Chadd, R., Extence, C., 2004. The conservation of freshwater macroinvertebrate populations: a community-based classification scheme. Aquat. Conserv.: Mar. Freshwater Ecosyst. 14, 597–624. http://dx.doi.org/10.1002/aqc.630

Chandesris, A., Mengin, N., Malavoi, J. R., Souchon, Y., Pella, H., & Wasson, J. G., 2008. Système Relationnel d'Audit de l'Hydromorphologie des Cours d'Eau. Principes et methodes, v3, 1, 81

Chapin, T. P., Todd, A. S., & Zeigler, M. P., 2014. Robust, low-cost data loggers for stream temperature, flow intermittency, and relative conductivity monitoring. Water Resources Research, 50(8), 6542-6548. https://doi.org/10.1002/2013WR015158

Chiu, M.-C., Leigh, C., Mazor, R., Cid, N., Resh, V., 2017. Anthropogenic threats to intermittent rivers and ephemeral streams. In: Datry, T., Bonada, N., Boulton, A.J. (Eds.), Intermittent Rivers and Ephemeral Streams: Ecology and Management. Elsevier, Amsterdam, the Netherlands, pp. 433–454. http://dx.doi.org/10.1016/B978-0-12-803835-2.00017-6

Cid, N., Bonada, N., Carlson, S., Grantham, T., Gasith, A., Resh, V., 2017. High variability is a defining component of Mediterranean-climate rivers and their biota. Water 9, 52. https://doi.org/10.3390/w9010052

Cid, N., Bonada, N., Heino, J., Cañedo-Argüelles, M., Crabot, J., Sarremejane, R., Soininen, J., Stubbington, R., Datry, T. Submitted. A metacommunity approach to improve biological assessments in highly dynamic freshwater ecosystems.

Clavero M, Hermoso V., 2015. Historical data to plan the recovery of the European eel. Journal of Applied Ecology 52: 960-968. 10.1111/1365-2664.12446

Corti, R., Datry, T., 2016. Terrestrial and aquatic invertebrates in the riverbed of an intermittent river: parallels and contrasts in community organisation. Freshwater Biol. 61, 1308–1320. https://doi.org/10.1111/fwb.12692

Corti, R., Larned, S.T., Datry, T., 2013. A comparison of pitfall-trap and quadrat methods for sampling ground-dwelling invertebrates in dry riverbeds. Hydrobiologia 717, 13–26. https://doi.org/10.1007/s10750-013-1563-0

Coste, M., 1986. Les méthodes microfloristiques d'évaluation de la qualité des eaux. Cemagref, Bordeaux, 15 pp. + 46 annexes

Costigan, K. H., Daniels, M. D., & Dodds, W. K., 2015. Fundamental spatial and temporal disconnections in the hydrology of an intermittent prairie headwater network. Journal of Hydrology, 522, 305-316. https://doi.org/10.1016/j.jhydrol.2014.12.031

Costigan, K. H., Kennard, M. J., Leigh, C., Sauquet, E., Datry, T., & Boulton, A. J., 2017. Flow regimes in intermittent rivers and ephemeral streams. In Intermittent Rivers and Ephemeral Streams (pp. 51-78). Academic Press. https://doi.org/10.1016/B978-0-12-803835-2.00003-6

Crivelli, A.J. 2006. *Pelasgus laconicus*. The IUCN Red List of Threatened Species 2006: e.T61374A12457331. Available at: http://dx.doi.org/10.2305/IUCN.UK.2006.RLTS.T61374A12457331.en Accessed 29 July 19.

Crivelli, A.J. 2006. *Squalius keadicus*. The IUCN Red List of Threatened Species 2006: e.T60738A12402522. Available at: http://dx.doi.org/10.2305/IUCN.UK.2006.RLTS.T60738A12402522.en Accessed 29 July 19.

Crivelli, A.J. 2006. *Tropidophoxinellus spartiaticus*. The IUCN Red List of Threatened Species 2006: e.T22346A9370791. Available at: http://dx.doi.org/10.2305/IUCN.UK.2006.RLTS.T22346A9370791.en Accessed 29 July 19.

Croke, B., Jakeman, A., 2007. Use of the IHACRES rainfall-runoff model in arid and semi-arid regions. In H. Wheater, S. Sorooshian, & K. Sharma (Eds.), Hydrological Modelling in Arid and Semi-Arid Areas (International Hydrology Series, pp. 41-48). Cambridge: Cambridge University Press. doi:10.1017/CBO9780511535734.005

D'Ambrosio E, De Girolamo AM, Barca E, Ielpo P, Rulli M (2017). Characterising the hydrological regime of an ungauged temporary river system: a case study. *Environmental Science and Pollution Research*, **24**, Issue 16, 13950–13966. https://doi.org/10.1007/s11356-016-7169-0.

Dahm, C.N., Baker, M.A., Moore, D.I., Thibault, J.R., 2003. Coupled biogeochemical and hydrological responses of streams and rivers to drought. Freshwater Biol. 48, 1219-1231. http://doi.org/10.1046/j.1365-2427.2003.01082.x

Daniel, E. B., Camp, J. V., LeBoeuf, E. J., Penrod, J. R., Dobbins, J. P., & Abkowitz, M. D., 2011. Watershed modeling and its applications: A state-of-the-art review. The Open Hydrology Journal, 5(1). https://doi.org/10.2174/1874378101105010026

Datry, T., Corti, R., Philippe, M., 2012. Spatial and temporal aquatic–terrestrial transitions in the temporary Albarine River, France: responses of invertebrates to experimental rewetting. Freshwater Biol., 57, 716–727. https://doi.org/10.1111/j.1365-2427.2012.02737.x

Datry, T., Larned, S.T., Tockner, K., 2014a. Intermittent rivers: a challenge for freshwater ecology. BioScience 64, 229–235. https://doi.org/10.1093/biosci/bit027

Datry, T., Larned, S.T., Fritz, K.M., Bogan, M.T., Wood, P.J., Meyer, E.I., Santos, A.N., 2014b. Broad-scale patterns of invertebrate richness and community composition in temporary rivers: effects of flow intermittence. Ecography 37, 94–104. https://doi.org/10.1111/j.1600-0587.2013.00287.x

Datry, T., Bonada, N., Heino, J., 2016. Towards understanding the organisation of metacommunities in highly dynamic ecological systems. Oikos 125, 149–159. https://doi.org/10.1111/oik.02922

Datry, T., Bonada, N., Boulton, A.J., 2017a. General introduction. In: Datry, T., Bonada, N., Boulton, A.J. (Eds.), Intermittent Rivers and Ephemeral Streams: Ecology and Management. Amsterdam, the Netherlands, Elsevier, pp. 1–20. https://doi.org/10.1016/B978-0-12-803835-2.00001-2

Datry, T., Boulton, A.J., Bonada, N., Fritz, K., Leigh, C., Sauquet, E., Tockner, K., Hugueny, B., Dahm, C.N., 2018. Flow intermittence and ecosystem services in rivers of the Anthropocene. J. Appl. Ecol. 55, 353–364. DOI: 10.1111/1365-2664.12941

Datry, T., Singer, G., Sauquet, E., Jorda-Capdevila, D., von Schiller, D., Stubbington, R., Magand, C., Pařil, P., Miliša, M., Acuña, V., Alves, H. M., Augeard, B., Brunke, M., Cid, N., Csabai, Z., England, J., Froebrich, J., Koundouri, P., Lamouroux, N., Martí, E., Morais, M., Munné, A., Mutz, M., Pesic, V., Previšić, A., Reynaud, A., Robinson, C., Sadler, J., Skoulikidis, N., Terrier, B., Tockner, K., Vesely, D., Zoppini, A., 2017b. Science and management of intermittent rivers and ephemeral streams (SMIRES). Research Ideas and Outcomes 3: e21774. https://doi.org/10.3897/rio.3.e21774

Davis, S., Golladay, S. W., Vellidis, G., & Pringle, C. M., 2003. Macroinvertebrate biomonitoring in intermittent coastal plain streams impacted by animal agriculture. Journal of Environmental Quality, 32(3), 1036-1043. doi:10.2134/jeq2003.1036

Dawson, F.H., Newman, J.R., Gravelle, M.J., Rouene K.J., Henville, P., 1999. Assessment of the trophic status of rivers using macrophytes: Evaluation of the mean trophic rank. R&D Technical Report E39. Environment Agency, Bristol, UK, 32 pp.

Daily, G. C. (1997). *Nature's services* (Vol. 19971). Island Press, Washington, DC.

Dean, J. F., Camporese, M., Webb, J. A., Grover, S. P., Dresel, P. E., Daly, E., 2016, Water balance complexities in ephemeral catchments with different land uses: Insights from monitoring and distributed hydrologic modeling, Water Resour. Res., 52, 4713–4729, doi:10.1002/2016WR018663

De Girolamo, A. M., Gallart, F., Pappagallo, G., Santese, G., & Porto, A. L., 2015. An eco-hydrological assessment method for temporary rivers. The Celone and Salsola rivers case study (SE, Italy). In Annales de Limnologie-International Journal of Limnology (Vol. 51, No. 1, pp. 1-10). EDP Sciences.

De Girolamo, A M, Lo Porto, A., Pappagallo, G., Tzoraki, O., Gallart, F., 2015. The hydrological status concept: application at a temporary river (Candelaro, Italy). River Res. Appl. 31, 892–903. https://doi.org/10.1002/rra.2786

De Girolamo, A.M., Barca, E., Ielpo, P., Rulli, M.C., 2016. Characterising the hydrological regime of an ungauged temporary river system: a case study. Environmental Science and Pollution Research, 1-17.

De Girolamo, A.M., Bouraoui, F., Buffagni, A., Pappagallo, G., & Lo Porto, A., 2017a. Hydrology under climate change in a temporary river system: Potential impact on water balance and flow regime. River Research and Applications, 33, 1219– 1232. https://doi.org/10.1002/rra.3165

De Girolamo A, Barca E, Pappagallo E, Lo Porto A, 2017b. Simulating ecologically relevant hydrological indicators in a temporary river system. *Agricultural Water Management*, **180** (Part B), 194-204, doi: DOI10.1016/j.agwat.2016.05.034

De Sostoa, A., Caiola, N., Casals, F., García-Berthou, E., Alcaraz, C., Benejam, L., Maceda, A., Solà, C., Munné, A., 2010. Adjustment of the index of biotic integrity (IBICAT) based on the use of fish as indicators of the environmental quality of the rivers of Catalonia (in Catalan). Agència Catalana de l'Aigua, Departament de Medi Ambient i Habitatge, Generalitat de Catalunya, Barcelona, 187 p. http://doi.org/10.13140/2.1.1551.6964.

Deil, U., 2005. A review on habitats, plant traits and vegetation of ephemeral wetlands – a global perspective. Phytocoenologia 35, 533–705. https://dx.doi.org/10.1127/0340-269X/2005/0035-0533

Del Rosario, R.B., Resh, V.H., 2000. Invertebrates in intermittent and perennial streams: is the hyporheic zone a refuge from drying? J. N. Am. Benthol. Soc. 19, 680–696. https://doi.org/10.1899/08-017.1

Dida, G. O., Anyona, D. N., Abuom, P. O., Akoko, D., Adoka, S. O., Matano, A. S., ... & Ouma, C. (2018). Spatial distribution and habitat characterization of mosquito species during the dry season along the Mara River and its tributaries, in Kenya and Tanzania. Infectious diseases of poverty, 7(1), 2. https://doi.org/10.1186/s40249-017-0385-0

Dodds, W.K., Gido, K., Whiles, M.R., Fritz, K.M., Matthews, W.J., 2004. Life on the edge: the ecology of Great Plains prairie streams. Bioscience 54, 205–216. https://doi.org/10.1641/0006-3568(2004)054%5b0205:LOTETE%5d2.0.CO;2

Dolédec, S., Statzner, B., Bournard, M., 1999. Species traits for future biomonitoring across ecoregions: patterns along a human-impacted river. Freshwater Biol. 42, 737–758. https://doi.org/10.1046/j.1365-2427.1999.00509.x

Döll, P., 2009. Vulnerability to the impact of climate change on renewable groundwater resources: a global-scale assessment. Environ. Res. Lett. 4 035006.

Döll, P. and Zhang, J., 2010 Impact of climate change on freshwater ecosystems: a global-scale analysis of ecologically relevant river flow alterations. Hydrology and Earth System Sciences, 14, 783–799.

Döll, P., & Schmied, H. M., 2012. How is the impact of climate change on river flow regimes related to the impact on mean annual runoff? A global-scale analysis. Environ. Res. Lett., 7, 014037. https://doi.org/10.1088/1748-9326/7/1/014037

EEA, 2018. European waters Assessment of status and pressures 2018. EEA Report No 7/2018.

Elbrecht, V., Leese, F., 2015. Can DNA-based ecosystem assessments quantify species abundance? Testing primer bias and biomass-sequence relationships with an innovative metabarcoding protocol. PLoS One 10, e0130324. https://doi.org/10.1371/journal.pone.0130324

Elbrecht, V., Vamos, E.E., Meissner, K., Aroviita, J., Leese, F., 2017. Assessing strengths and weaknesses of DNA metabarcoding-based macroinvertebrate identification for routine stream monitoring. Methods Ecol. Evol. 8, 1265–1275. https://doi.org/10.1111/2041-210X.12789

Elbrecht, V., Vamos, E.E., Steinke, D., Leese, F., 2018. Estimating intraspecific genetic diversity from community DNA metabarcoding data. PeerJ 6, e4644. https://doi.org/10.7717/peerj.4644

Elosegi, A., Díez, J., Mutz, M., 2010. Effects of hydromorphological integrity on biodiversity and functioning of river ecosystems. Hydrobiol. 657**,** 199–215. http://doi.org/10.1007/s10750-009-0083-4

England, J., Chadd, R., Dunbar, M.J., Sarremejane, R., Stubbington, R., Westwood, C.G., Leeming, D., 2019. An invertebrate-based index to characterize ecological responses to flow intermittence in rivers. Fundam. Appl. Limnol*.* http://dx.doi.org/10.1127/fal/2019/1206

European Commission, 2000. Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy.

European Commission (EC), 2003. Common Implementation Strategy for the Water Framework Directive (2000/60/EC). Guidance Document No. 11: Planning Processes. Office for Official Publications of the European Communities, Luxembourg. http://ec.europa.eu/environment/water/water-framework/facts_figures/guidance_docs_en.htm Accessed 10 July 19.

European Commission, 2009. Common Implementation Strategy for the Water Framework Directive (2000/60/EC). Guidance document (23). Available at: http://ec.europa.eu/environment/water/water-framework/objectives/pdf/strategy2.pdf

European Commission, 2015. Ecological flows in the implementation of the Water Framework Directive. Guidance Document No. 31. Technical Report - 2015 - 086.ISBN. 106 pp. 978-92-79-45758-6. ISSN 1725-1087. DOI: 10.2779/775712

European Commission, 2018. Commission Decision (EU) 2018/229 of 12 February 2018 establishing, pursuant to Directive 2000/60/EC of the European Parliament and of the

Council, the values of the Member State monitoring system classifications as a result of the intercalibration exercise and repealing Commission Decision 2013/480/EU.

European Commission, 2019. Fitness Check of the Water Framework Directive and the Floods Directive. European Commission. SWD(2019) 439 final. 176 pp.

Extence C., Balbi D., Chadd R., 1999. River flow indexing using British benthic macroinvertebrates: A framework for setting hydroecological objectives. Regul. Rivers: Res. Manage. 15, 543–574. https://doi.org/10.1002/(SICI)1099-1646(199911/12)15:6<545::AID-RRR561>3.0.CO;2-W

Falasco, E., Piano, E., Bona, F., 2016a. Diatom flora in Mediterranean streams: flow intermittency threatens endangered species. Biodivers. Conserv. 25, 2965–2986. https://doi.org/10.1007/s10531-016-1213-8

Falasco, E., Piano, E., Bona, F., 2016b. Suggestions for diatom-based monitoring in intermittent rivers. Knowl. Manag. Aquat. Ecosyst. 417, 38. http://dx.doi.org/10.1051/kmae/2016025

FAME Consortium. 2004. Manual for the application of the European Fish Index - EFI. A fish-based method to assess the ecological status of European rivers in support of the Water Framework Directive. Version 1.1, January 2005. Available at: https://fame.boku.ac.at/downloads/manual_Version_Februar2005.pdf Accessed 29 July 19

Ferreira, T., Caiola, N., Casals, F., Oliveira, J.M., De Sostoa, A., 2007b. Assessing perturbation of river fish communities in the Iberian Ecoregion. Fisheries Manag. Ecol. 14, 519–530. https://doi.org/10.1111/j.1365-2400.2007.00581.x

Ferreira, T., Oliveira, J., Caiola, N., De Sostoa, A., Casals, F., Cortes, R., Economou, A., Zogaris, S., Garcia-Jalon, D., Ilhéu, M., Martinez-Capel, F., 2007a. Ecological traits of fish assemblages from Mediterranean Europe and their responses to human disturbance. Fisheries Manag. Ecol. 14, 473–481. https://doi.org/10.1111/j.1365-2400.2007.00584.x

Fritz, K. M., Feminella, J. W., Colson, C., Lockaby, B. G., Governo, R., & Rummer, R. B., 2006. Biomass and decay rates of roots and detritus in sediments of intermittent Coastal Plain streams. Hydrobiologia, 556(1), 265-277. https://doi.org/10.1007/s10750-005-1154-9

Fritz, K.M., Cid, N., Autrey, B., 2017. Governance, legislation and protection of intermittent rivers and ephemeral streams. In: Datry, T., Bonada, N., Boulton, A.J. (Eds.), Intermittent Rivers and Ephemeral Streams: Ecology and Management. Elsevier, Amsterdam, the Netherlands, pp. 477–508. https://doi.org/10.1016/B978-0-12-803835-2.00019-X

Fryirs, K., & Brierley, G., 2005. Practical application of the River Styles Framework as a tool for catchment-wide river management: a case study from bega catchment, new south wales. Auckland: Macquirie University, 227

Gallart, F., Prat i Fornells, N., García-Roger, E. M., Latron, J., Rieradevall i Sant, M., Llorens, P., ... & Buffagni, A., 2012. A novel approach to analysing the regimes of temporary streams in relation to their controls on the composition and structure of aquatic biota. Hydrology and Earth System Sciences, 2012, vol. 16, p. 3165-3182. http://dx.doi.org/10.5194/hess-16-3165-2012

Gallart, F., Llorens, P., Latron, J., Cid, N., Rieradevall, M., & Prat, N., 2016. Validating alternative methodologies to estimate the regime of temporary rivers when flow data are unavailable. Science of the Total Environment, 565, 1001-1010. https://doi.org/10.1016/j.scitotenv.2016.05.116

Gallart, F., Cid, N., Latron, J., Llorens, P., Bonada, N., Jeuffroy, J., Jiménez-Argudo, S.M., Vega, R.M., Solà, C., Soria, M., Bardina, M., Hernández-Casahuga, A.-J., Fidalgo, A., Estrela, T., Munné, A., Prat, N., 2017. TREHS: An open-access software tool for investigating and evaluating temporary river regimes as a first step for their ecological status assessment. Sci. Total Environ. 607, 519–540. https://doi.org/10.1016/j.scitotenv.2017.06.209

Gamvroudis, C., Nikolaidis, N.P., Tzoraki, O., Papadoulakis, V., Karalemas, N., 2015. Water and sediment transport modeling of a large temporary river basin in Greece. Sci. Total Environ. 508. https://doi.org/10.1016/j.scitotenv.2014.12.005

García-Berthou, E., Bae, M.-J., Benejam, L., Alcaraz, C., Casals, F., de Sostoa, A., Solà, C., Munné, A., 2015. Fish-based indices in Catalan rivers: intercalibration and comparison of approaches. In: Munné, A., Ginebreda, A., Prat, N. (Eds.), Experiences from Surface Water Quality Monitoring: The Handbook of Environmental Chemistry, 42. Springer, Cham, pp. 125–147. https://doi.org/10.1007/698_2015_342

García-Comendador J, Fortesa J, Calsamiglia A, Calvo-Cases A, Estrany J. 2017. Post-fire hydrological response and suspended sediment transport of a terraced Mediterranean catchment. Earth Surface Processes and Landforms 42 : 2254–2265. DOI: 10.1002/esp.4181 [online] Available from: https://onlinelibrary.wiley.com/doi/full/10.1002/esp.4181

Gassman, P., Reyes, M., Green, C. & Arnold, J., 2007. The Soil and Water Assessment Tool: historical development, applications, and future research directions. Trans. Am. Soc. Agric. Biol. Engrs 50(4), 1211–1250.

Gavrilovic, Z., 1988. “The Use of Empirical Method (Erosion Potential Method) for Calculating Sediment Production and Transportation in Unstudied or Torrential Streams.” In Proceedings of the International Conference of River Regime, 18–20 May, Wallingford, England, pp. 411–422.

Gionchetta, G., Oliva, F., Menéndez, M., Lopez Laseras, P., Romaní, A.M., 2019. Key role of streambed moisture and flash storms for microbial resistance and resilience to long-term drought. Freshwater Biol 64, 306-322. http://doi.org/10.1111/fwb.13218

Gómez, R., Arce, M.I., Baldwin, D.S., Dahm, C.N., 2017. Water Physicochemistry in Intermittent Rivers and Ephemeral Streams. In Intermittent Rivers and Ephemeral Streams, eds. T. Datry, N. Bonada, A. Boulton, 109-134. Academic Press.

González del Tánago, M., García de Jalón, D., 2011. Riparian Quality Index (RQI): A methodology for characterising and assessing the environmental conditions of riparian zones. Limnetica 30, 235–254. https://doi.org/10.23818/limn.30.18

Goodrich, D.C., Kepner, W.G., Levick, L.R., Wigington Jr, P.J., 2018. Southwestern Intermittent and Ephemeral Stream Connectivity. JAWRA J. Am. Water Resour. Assoc. 54, 400–422. https://doi.org/10.1111/1752-1688.12636

Gosling, S. N., Taylor, R. G. Arnell, N. W. and Todd, M. C., 2011. A comparative analysis of projected impacts of climate change on river runoff from global and catchment-scale hydrological models, Hydrol. Earth Syst. Sci. 15 279–94. https://doi.org/10.5194/hess-15-279-2011

Götz, R., Steiner, B., Friesel, P., Roch, K., Walkow, F., Maaβ, V., 1998. Dioxin (PCDD/F) in the river elbe - investigations of their origin by multivariate statistical methods. Chemosphere. 37, 1987-2002. http://doi.org//10.1016/S0045-6535(98)00263-X

Grill, G., Lehner, B., Thieme, M., Geenen, B., Tickner, D., Antonelli, F., Babu, S., Borrelli, P., Cheng, L., Crochetiere, H., 2019. Mapping the world’s free flowing rivers. Nature 569, 215. https://doi.org/10.1038/s41586-019-1111-9

Gudmundsson, L., Leonard, M., Do, H.X., Westra, S., Seneviratne, S.I., 2019. Observed trends in global indicators of mean and extreme streamflow. Geophys. Res. Lett. 46, 756–766. https://doi.org/10.1029/2018GL079725

Gurnell, A. M., Rinaldi, M., Belletti, B., Bizzi, S., Blamauer, B., Braca, G., & Demarchi, L., 2016. A multi-scale hierarchical framework for developing understanding of river behaviour to support river management. Aquatic Sciences, 78(1), 1-16. https://doi.org/10.1007/s00027-015-0424-5

Gutiérrez-Cánovas, C., Arribas, P., Naselli-Flores, L., Bennas, N., Finocchiaro, M., Millán, A., Velasco, J., 2019. Evaluating anthropogenic impacts on naturally stressed ecosystems: Revisiting river classifications and biomonitoring metrics along salinity gradients. Sci. Total Environ. 658, 912–921. https://doi.org/10.1016/j.scitotenv.2018.12.253

Hajibabaei, M., Shokralla, S., Zhou, X., Singer, G.A., Baird, D.J., 2011. Environmental barcoding: a next-generation sequencing approach for biomonitoring applications using river benthos. PLoS One 6, e17497. https://doi.org/10.1371/journal.pone.0017497

Hakenkamp, C., Palmer, M.A., 2000. The ecology of hyporheic meiofauna. In: Jones, J.B., Mullholland, P.J. (Eds.), Streams and Groundwaters. Academic Press, London, pp. 307–336. https://doi.org/10.1016/B978-012389845-6/50014-4

Hassan, M.A., Egozi, R., 2001. Impact of wastewater discharge on the channel morphology of ephemeral streams. Earth Surf. Process. Landf. 26, 1285–1302. https://doi.org/10.1002/esp.273

Hebert, P.D., Cywinska, A., Ball, S.L., Dewaard, J.R., 2003. Biological identifications through DNA barcodes. Proc. Roy. Soc. B-Biol. Sci. 270, 313–321. https://doi.org/10.1098/rspb.2002.2218

Heggenes, J. & Wollebaek, J., 2013 Habitat Use and Selection by Brown Trout in Streams. Ecohydraulics: an integrated approach (eds I. Maddock, A. Harby, P. Kemp, & P.J. Wood), p. 462. John Wiley & Sons, Ltd, Chichester, United Kingdom. https://doi.org/10.1002/9781118526576.ch9

Heino, J., 2013. The importance of metacommunity ecology for environmental assessment research in the freshwater realm. Biol. Rev. 88, 166–178. https://doi.org/10.1111/j.1469-185X.2012.00244.x

Heino, J., Alahuhta, J., Ala-Hulkko, T., Antikainen, H., Bini, L.M., Bonada, N., Datry, T., Erős, T., Hjort, J., Kotavaara, O., Melo, A.S., Soininen, J., 2017. Integrating dispersal proxies in ecological and environmental research in the freshwater realm. Environ. Rev. 25, 334–349. https://doi.org/10.1139/er-2016-0110

Herczeg, A.L., Dogramaci, S.S., Leaney, F.W.J., 2001. Origin of dissolved salts in a large, semi-arid groundwater system: Murray Basin, Australia. Mar. Freshwater. Res. 52, 41-52. http://doi.org/10.1071/MF00040

Hering, D., Johnson, R.K., Kramm, S., Schmutz, S., Szoszkiewicz, K., Verdonschot. P.F.M., 2006. Assessment of European streams with diatoms, macrophytes, macroinvertebrates and fish: a comparative metric-based analysis of organism response to stress. Freshwater Biol 51, 1757-1785. http://doi.org/10.1111/j.1365-2427.2006.01610.x

Hermoso, V., Clavero, M., Blanco-Garrido, F., Prenda, J., 2010. Assessing the ecological status in species-poor systems: a fish-based index for Mediterranean Rivers (Guadiana River, SW Spain). Ecol. Indic. 10, 1152–1161. https://doi.org/10.1016/j.ecolind.2010.03.018

Hisdal, H., Stahl, K., Tallaksen, L.M. and Demuth, S., 2001. Have streamflow droughts in Europe become more severe or frequent? International Journal of Climatology 21: 317–333.https://sustainabledevelopment.un.org/

Hladyz, S., Watkins, S.C., Whitworth, K.L., Baldwin, D.S., 2011. Flows and hypoxic blackwater events in managed ephemeral river channels. J. Hydrol. 401**,** 117-125. http://doi.org/10.1016/j.jhydrol.2011.02.014

Hoerling, M., Eischeid, J., Perlwitz, J., Quan, X., Zhang, T., Pegion, P., 2011. On the Increased Frequency of Mediterranean Drought. J. Climate. 25**,** 2146-2161. http://doi.org/10.1175/jcli-d-11-00296.1

Holmes, N.T.H., Newman, J.R., Chadd, S., Rouen, K.J., Saint, L., Dawson, F.H., 1999. Mean trophic rank: A user's manual. Environment Agency R&D Technical Report E38. 132 pp. Available at: https://www.gov.uk/government/publications/mean-trophic-rank-a-user-manual Accessed 29 July 19

Hughes, S.J., Santos, J.M., Ferreira, M.T., Caraca, R., Mendes, A.M., 2009. Ecological assessment of an intermittent Mediterranean river using community structure and function: evaluating the role of different organism groups. Freshwater Biol. 54, 2383–2400. https://doi.org/10.1111/j.1365-2427.2009.02253.x

Hughes, S.J., Santos, H.M., Ferreira, M.T., Mendes, A., 2010. Evaluating the response of biological assemblages as potential indicators for restoration measures in an intermittent Mediterranean river. Environ. Manage. 46, 285–301. https://doi.org/10.1007/s00267-010-9521-3

INAG (Instituto da Água), 2009. Critérios para a classificação do estado das massas de água superficiais – rios e albufeiras. INAG. Ministério do Ambiente, do Ordenamento do Território e do Desenvolvimento Regional, Lisbon, Portugal, 71 p. Available at: https://apambiente.pt/dqa/assets/crit%C3%A9rios-classifica%C3%A7%C3%A3o-rios-e-albufeiras.pdf Accessed 28 July 19.

Ivkovic, K. M., Croke, B. F. W., & Kelly, R. A., 2014. Overcoming the challenges of using a rainfall–runoff model to estimate the impacts of groundwater extraction on low flows in an ephemeral stream. Hydrology Research, 45(1), 58-72. doi: 10.2166/nh.2013.204

Jaeger, K. L., & Olden, J. D., 2012. Electrical resistance sensor arrays as a means to quantify longitudinal connectivity of rivers. River Research and Applications, 28(10), 1843-1852. https://doi.org/10.1002/rra.1554

Johnson, M. F., Hannah, C., Acton, L., Popovici, R., Karanth, K. K., & Weinthal, E., 2014. Network environmentalism: Citizen scientists as agents for environmental advocacy. Global Environmental Change, 29, 235-245. https://doi.org/10.1016/j.gloenvcha.2014.10.006

Jorda-Capdevila, D., Iniesta-Arandia, I., Quintas-Soriano, C., Basdeki, A., Calleja, E., De Girolamo, A.M., Gilvear, M., Ilhéu, M., Kriaučiūniene, J., Logar, I., Loures, L, and Padło, T. (submitted). Disentangling the complexity of social values of temporary rivers. *Ecosystems and People*.

Jorda-Capdevila, D., & Rodríguez-Labajos, B., 2015. An ecosystem service approach to understand conflicts on river flows: local views on the Ter River (Catalonia). Sustainability Science, 10(3), 463-477. https://doi.org/10.1007/s11625-014-0286-0

Jorda-Capdevila, D., & Rodríguez-Labajos, B., 2017a. Socioeconomic value (s) of restoring environmental flows: systematic review and guidance for assessment. *River Research and Applications*, *33*(3), 305-320. https://doi.org/10.1002/rra.3074

Jorda-Capdevila, D., & Rodríguez-Labajos, B., 2017b. Embracing complexity improves the assessment of environmental flows–One step beyond Gopal's (2016) framework. *Ecosystem services*, *25*, 79-81. https://doi.org/10.1016/j.ecoser.2017.03.018

Jorde K., Chneider, M., Peter, A. & Zöllner, F., 2001 Models for the evaluation of fish habitat quality and instream flow assessment. International Symposium on Environmental Hydraulics, p. Tempe, U.S.A.

Kahlert, M., Rimet, F., Bouchez, A., Kelly, M., Sato, S., Mann, D., 2019. Connecting the morphological and molecular species concepts to facilitate species identification within the genus Fragilaria (Bacillariophyta). J. Phycol. 55, 948–970 https://doi.org/10.1111/jpy.12886

Kail, J., Brabec, K., Poppe, M., Januschke, K., 2015. The effect of river restoration on fish, macroinvertebrates and aquatic macrophytes: A meta-analysis. Ecol. Indic. 58, 311–321. http://dx.doi.org/10.1016/j.ecolind.2015.06.011

Kalogianni, E., Vourka, A., Karaouzas, I., Vardakas, L., Laschou, S., Skoulikidis, N.T., 2016. Combined effects of water stress and pollution on macroinvertebrate and fish assemblages in a Mediterranean intermittent river. Sci. Total Environ. 603, 639–650. https://doi.org/10.1016/j.scitotenv.2017.06.078

Karaouzas, I., Theodoropoulos, C., Vardakas, L., Kalogianni, E., Th. Skoulikidis, N., 2018. A review of the effects of pollution and water scarcity on the stream biota of an intermittent Mediterranean basin. River Res. Appl. 34, 291–299. https://doi.org/10.1002/rra.3254

Karydas, C.G., Panagos, P., 2016. Modelling monthly soil losses and sediment yields in Cyprus. Int. J. Digit. Earth 9, 766–787. https://doi.org/10.1080/17538947.2016.1156776

Karydas, C.G., Panagos, P., 2018. The G2 erosion model: An algorithm for month-time step assessments. Environ. Res. 161, 256–267. https://doi.org/10.1016/j.envres.2017.11.010

Katz, G.L., Denslow, M.W., Stromberg, J.C., 2012. The Goldilocks effect: intermittent streams sustain more plant species than those with perennial or ephemeral flow. Freshwater Biol. 57, 467–480. https://doi.org/10.1111/j.1365-2427.2011.02714.x

Kells, A. R., & Goulson, D., 2003. Preferred nesting sites of bumblebee queens (Hymenoptera: Apidae) in agroecosystems in the UK. Biological conservation, 109(2), 165-174. https://doi.org/10.1016/S0006-3207(02)00131-3

Kelly, M.G., Whitton, B.A., 1995. The trophic diatom index: a new index for monitoring eutrophication in rivers. J. Appl. Phycol. 7, 433–444. https://doi.org/10.1007/BF00003802

Kerezsy, A., Gido, K., Magalhães, M.F. Skelton, P.H., 2017. The biota of intermittent rivers and ephemeral streams: Fishes. In: Datry, T., Bonada, N., Boulton, A.J. (Eds.), Intermittent Rivers and Ephemeral Streams: Ecology and Management. Elsevier, Amsterdam, the Netherlands, pp. 273–298. https://doi.org/10.1016/B978-0-12-803835-2.00010-3

Kermarrec, L., Franc, A., Rimet, F., Chaumeil, P., Humbert, J.F., Bouchez, A., 2013. Next-generation sequencing to inventory taxonomic diversity in eukaryotic communities: a test for freshwater diatoms. Mol. Ecol. Resour. 13, 607–619. https://doi.org/10.1111/1755-0998.12105

Kolkwitz, R., Marsson M., 1909. Ökologie der tierischen Saprobien. Beiträge zur Lehre von der biologischen Gewässerbeurteilung. Int. Rev. ges. Hydrobiol., 2, 126–152. https://doi.org/10.1002/iroh.19090020108

Koundouri, P. 2015, The Use of Ecosystem Services Approach in Guiding Water Valuation and Management: Inland and Coastal Waters. In Dinar, A. and Schwabe, K. (eds.) *Handbook of Water Economics* (pp. 126-149), Edward Elgar Publishers Ltd.

Koundouri, P., Boulton, A. J., Datry, T., & Souliotis, I., 2017. Ecosystem services, values, and societal perceptions of intermittent rivers and ephemeral streams. In *Intermittent rivers and ephemeral streams* (pp. 455-476). Academic Press.

Koundouri, P., Akinsete, E., & Tsani, S., 2019. Socio-economic and policy implications of multi-stressed rivers: a European perspective. In *Multiple Stressors in River Ecosystems* (pp. 335-351). Elsevier.

Koutrakis E.T. , Triantafillidis S, Sapounidis A.S., Vezza P. , Kamidis N., Sylaios G., Comoglio C. 2018. Evaluation of ecological flows in highly regulated rivers using the mesohabitat approach: A case study on the Nestos River, N. Greece. Ecohydrology & Hydrobiology . https://doi.org/10.1016/j.ecohyd.2018.01.002

Knighton, A. D., & Nanson, G. C., 2001. An event-based approach to the hydrology of arid zone rivers in the Channel Country of Australia. Journal of Hydrology, 254(1-4), 102-123. https://doi.org/10.1016/S0022-1694(01)00498-X

Laliberté, E., Wells, J.A., DeClerck, F., Metcalfe, D.J., Catterall, C.P., Queiroz, C., Aubin, I., Bonser, S.P., Ding, Y., Fraterrigo, J.M., McNamara, S., Morgan, J.W., Sánchez Merlos, D., Vesk, P.A., Mayfield, M.M., 2010. Land-use intensification reduces functional redundancy and response diversity in plant communities. Ecol. Lett. 13, 76–86. https://doi.org/10.1111/j.1461-0248.2009.01403.x

Larned, S.T., Datry, T., Arscott, D.B., Tockner, K., 2010. Emerging concepts in temporary-river ecology. Freshwater Biol. 55, 717–738. https://doi.org/10.1111/j.1365-2427.2009.02322.x

Larned, S.T., Datry, T., Robinson, C.T., 2007. Invertebrate and microbial responses to inundation in an ephemeral river reach in New Zealand: effects of preceding dry periods. Aquat. Sci. 69, 554–567. https://doi.org/10.1007/s00027-007-0930-1

Lazaridou, M., Ntislidou, C., Karaouzas, I., Skoulikidis, N., 2016. Report development of a national assessment method for the ecological status of rivers in Greece, using the biological quality element "benthic macroinvertebrates", the Hellenic Evaluation System-2 (HESY-2), and harmonisation with the results of the completed intercalibration of the Med GIG (RM1, RM2, RM4, RM5). https://circabc.europa.eu/sd/a/a089d466-d417-4892-8208-934aca7d9530/GR_MZB_Intercalibration%20rivers_Greece_FINAL_ACCEPTED_OCT2016.PDF (accessed 29.05.17).

Lazaridou, M., Ntislidou, C., Karaouzas, I., Skoulikidis, N., 2018. Harmonisation of a new assessment method for estimating the ecological quality status of Greek running waters. Ecol. Indic. 84, 683–694. https://doi.org/10.1016/j.ecolind.2017.09.032

Leese, F., Bouchez, A., Abarenkov, K., Altermatt, F., Borja, Á., Bruce, K., Ekrem, T., Ciampor Jr., F., Ciamporová-Zaťovičová, Z., Costa, F.O., Duarte, S., Elbrecht, V., Fontanetokk, D., Franc, A., Geiger, M.F., Hering, D., Kahlert, M., Kalamuji Stroil, B., Kelly, M., Keskin, E., Liska, I., Mergen, P., Meissner, K., Pawlowski, J., Penev, L., Reyjol, Y., Rotter, A., Steinke, D., van der Wal, B., Vitecek, S., Zimmermann, J., Weigand, A.M., 2018. Why we need sustainable networks bridging countries, disciplines, cultures and generations for aquatic biomonitoring 2.0: a perspective derived from the DNAqua-Net COST action. Adv. Ecol. Res. 58, 63–99. https://doi.org/10.1016/bs.aecr.2018.01.001

Leibold, M.A., Holyoak, M., Mouquet, N., Amarasekare, P., Chase, J.M., Hoopes, M.F., Holt, R. D., Shurin, J.B., Law, R., Tilman, D., Loreau, M., Gonzalez, A., 2004. The metacommunity concept: a framework for multi-scale community ecology. Ecol. Lett. 7, 601–613. https://doi.org/10.1111/j.1461-0248.2004.00608.x

Leigh, C., Datry, T., 2017. Drying as a primary hydrological determinant of biodiversity in river systems: A broad-scale analysis. Ecography 40, 487–499. https://doi.org/10.1111/ecog.02230

Leigh, C., Boulton, A.J., Courtwright, J.L., Fritz, K., May, C.L., Walker, R.H., Datry, T., 2016. Ecological research and management of intermittent rivers: an historical review and future directions. Freshwater Biol. 61, 1181–1199. https://doi.org/10.1111/fwb.12646

Leigh, C., Stubbington, R., Sheldon, F., Boulton, A.J., 2013. Hyporheic invertebrates as bioindicators of ecological health in temporary rivers. Ecol. Indic. 32, 62–73. https://dx.doi.org/10.1016/j.ecolind.2013.03.006

Lencioni, V., Spitale, D., 2015. Diversity and distribution of benthic and hyporheic fauna in different stream types on an alpine glacial floodplain. Hydrobiologia 751, 73–87. https://doi.org/10.1007/s10750-014-2172-2

Levick, L., Hammer, S., Lyon, R., Murray, J., Birtwistle, A., Guertin, D., Goodrich, D.C., Bledsoe, B., Laituri, M., 2018. An ecohydrological stream type classification of intermittent and ephemeral streams in the Southwestern United States. Journal of Arid Environments. 155:16-35.

Loiselle, S., Thornhill, I., Bailey, N., 2016. Citizen science: advantages of shallow versus deep participation. Front. Environ. Sci. 4. https://doi.org/10.3389/conf.FENVS.2016.01.00001

M.A., 2005. Millennium Assessment of Ecosystems. Current state and trends. Washington D.C., U.S.

Macher, J.N., Salis, R.K., Blakemore, K.S., Tollrian, R., Matthaei, C.D., Leese, F., 2016. Multiple-stressor effects on stream invertebrates: DNA barcoding reveals contrasting responses of cryptic mayfly species. Ecol. Indic. 61, 159–169. https://doi.org/10.1016/j.ecolind.2015.08.024

Macher, J.N., Vivancos, A., Piggott, J.J., Centeno, F.C., Matthaei, C.D., Leese, F., 2018. Comparison of environmental DNA and bulk-sample metabarcoding using highly degenerate cytochrome c oxidase I primers. Mol. Ecol. Resour. 18, 1456–1468. https://doi.org/10.1111/1755-0998.12940

Magalhães, M.F., Batalha, D.C., Collares-Pereira, M.J., 2002. Gradients in stream fish assemblages across a Mediterranean landscape: contributions of environmental factors and spatial structure. Freshwater Biol., 47, 1015–1031. https://doi.org/10.1046/j.1365-2427.2002.00830.x

Magalhães, M.F., Ramalho, C.E., Collares-Pereira, M.J., 2008. Assessing biotic integrity in a Mediterranean watershed: development and evaluation of a fish-based index. Fisheries Manag. Ecol. 15, 273–289. https://doi.org/10.1111/j.1365-2400.2008.00612.x

Magoulick, D.D., Kobza, R.M., 2003. The role of refugia for fishes during drought: a review and synthesis. Freshwater Biol., 48, 1186–1198. https://doi.org/10.1046/j.1365-2427.2003.01089.x

Makovinska, J., Hlubikova, D., 2014. Phytobenthos of the River Danube. In: Liska, I. (Ed.), The Danube River Basin. The Handbook of Environmental Chemistry, 39. Springer, Berlin, Heidelberg, pp. 317–340. https://doi.org/10.1007/698_2014_310

Marcé, R., Obrador, B., Gómez-Gener, L., Catalán, N., Koschorreck, M., Arce, M. I., ... & von Schiller, D., 2019. Emissions from dry inland waters are a blind spot in the global carbon cycle. Earth-science reviews, 188, 240-248. https://doi.org/10.1016/j.earscirev.2018.11.012

Marchi L., Comiti F., Crema S., Cavalli M., 2019. Channel control works and sediment connectivity in the European Alps. Science of The Total Environment, 668, 389-399. Doi: 10.1016/j.scitotenv.2019.02.4162

Marcus, W. A., & Fonstad, M. A., 2008. Optical remote mapping of rivers at sub-meter resolutions and watershed extents. Earth Surface Processes and Landforms: The Journal of the British Geomorphological Research Group, 33(1), 4-24. https://doi.org/10.1002/esp.1637

Marshall, J.C., Menke, N., Crook, D.A., Lobegeiger, J.S., Balcombe, S.R., Huey, J.A., Fawcett, J.H., Bond, N.R., Starkey, A.H., Sternberg, D., Linke, S., 2016. Go with the flow: the movement behaviour of fish from isolated waterhole refugia during connecting flow events in an intermittent dryland river. Freshwater Biol. 61, 1242–1258. https://doi.org/10.1111/fwb.12707

Martí, E., Aumatell, J., Godé, L., Poch, M., Sabater, F., 2004. Nutrient retention efficiency in streams receiving inputs from wastewater treatment plants. J. Environ. Qual. 33, 285-293. http://doi.org/10.2134/jeq2004.0285

Martí, E., Riera, J.L., Sabater, F., 2010. Effects of Wastewater Treatment Plants on Stream Nutrient Dynamics Under Water Scarcity Conditions. In Water Scarcity in the Mediterranean: Perspectives Under Global Change, eds. S. Sabater & D. Barceló, 173-195. Berlin, Heidelberg: Springer Berlin Heidelberg.

Martin-Ortega, J., Jorda-Capdevila, D., Glenk, K., & Holstead, K. L., 2015. 2 What defines ecosystem services-based approaches?. In Martin-Ortega, J., Ferrier, R. C., Gordon, I. J., & Khan, S. *Water ecosystem services: A global perspective*. UNESCO Publishing.

Mazor, R.D., Stein, E.D., Ode, P.R., Schiff, K., 2014. Integrating intermittent streams into watershed assessments: applicability of an index of biotic integrity. Freshwater Sci. 33, 459–474. https://doi.org/10.1086/675683

McGill, B.J., Enquist, B.J., Weiher, E., Westoby, M., 2006. Rebuilding community ecology from functional traits. Trends Ecol. Evol. 21, 178–185. https://doi.org/10.1016/j.tree.2006.02.002

Miccoli, F.P., Giustini, M., Bruni, P., Vignini, P., Pescosolido, M., Cicolani, B., Di Sabatino, A., 2006. La biodiversità e la qualità ambientale delle sorgenti dei Monti della Laga (Parco Nazionale Gran Sasso – Laga, Appennino centrale. Atti del XXXVI Congresso della Società Italiana di Biogeografia, 93.

Miccoli, F.P., Lombardo, P., Cicolani, B., 2013. Indicator value of lotic water mites (Acari: Hydrachnidia) and their use in macroinvertebrate-based indices for water quality assessment purposes. Knowl. Manag. Aquat. Ecosyst. 411, 8. https://doi.org/10.1051/kmae/2013075.

Mihaljević, Z., Kerovec, M., Mrakovčić, M., Plenković, A., Alegro, A., Primc-Habdija, B., 2011.Testing of biological methods for ecological status assessment (Water Framework Directive 2000/60/EC) in representative river basins of the Pannonian and Dinaric ecoregions (in Croatian). Faculty of Science, University of Zagreb, Croatia, 253 p.

Millán, A., Velasco, J., Gutiérrez-Cánovas, C., Arribas, P., Picazo, F., Sánchez-Fernández, D., et al., 2011). Mediterranean saline streams in southeast Spain: What do we know? J. Arid. Environ. 75, 1352-1359. http://doi.org/10.1016/j.jaridenv.2010.12.010

Mitchell, I. Reginster, M. Rounsevell, S. Sabaté, S. Sitch, B. Smith, J. Smith, P. Smith, M.T. Sykes, K. Thonicke, W. Thuiller, G. Tuck, S. Zaehle B. and Zierl, B., 2005 Ecosystem service supply and vulnerability to global change in Europe. Science, 310: 1333–1337. https://doi.org/10.1126/science.1115233

Molina, H. P., Navarro, A. M., Osorio, M. R., Canales, A. R., Zapata, J. C., & Sánchez, R. A., 2006. Social and irrigation water management issues in some water user's associations of the Low Segura River Valley (Alicante, Spain). Sustainable Irrigation Management, Technologies and Policies, 96, 205.

Mondy, C.P., Villeneuve, B., Archaimbault, V., Usseglio-Polatera, P., 2012. A new macroinvertebrate-based multimetric index (I_2M_2) to evaluate ecological quality of French wadeable streams fulfilling the WFD demands: A taxonomical and trait approach. Ecol. Indic. 18, 452–467. https://doi.org/10.1016/j.ecolind.2011.12.013

Montesantou, B., Ector, L., Daniilides, D., Hlubikova, D., Goma, J., Spathari, S., Kafouris, S., Kotzageorgis, G., Vayanou, F., Kirkos, G., 2008. Evaluation of the ecological status of running waters of Cyprus using diatoms as biological indicators (phytobenthos) – implementation of the Water Framework Directive 2000/60/EE – Final Intercalibration Report. Contract No. 22/2007

Mor, J.-R., Ruhí, A., Tornés, E., Valcárcel, H., Muñoz, I., Sabater, S., 2018. Dam regulation and riverine food-web structure in a Mediterranean river. Sci. Total Environ. 625, 301–310. https://doi.org/10.1016/j.scitotenv.2017.12.296

Morais, M., Pinto, P., Guilherme, P., Rosado, J., Antunes, I., 2004. Assessment of temporary streams: the robustness of metric and multimetric indices under different hydrological conditions. Hydrobiologia 516, 229–249. https://doi.org/10.1023/B:HYDR.0000025268.66163.32

Munné, A., Prat, N., 2009. Use of macroinvertebrate-based multimetric indices for water quality evaluation in Spanish Mediterranean rivers: an intercalibration approach with the IBMWP index. Hydrobiologia 628, 203–225. https://doi.org/10.1007/s10750-009-9757-1

Munné, A., Prat, N., 2011. Effects of Mediterranean climate annual variability on stream biological quality assessment using macroinvertebrate communities. Ecol. Indic. 11, 651–662. https://doi.org/10.1016/j.ecolind.2010.09.004

Muñoz, I., Abril, M., Casas-Ruiz, J.P., Casellas, M., Gómez-Gener, L., Marcé, R., et al., 2018. Does the severity of non-flow periods influence ecosystem structure and function of temporary streams? A mesocosm study. Freshw. Biol. 63, 613-625. http://doi.org/10.1111/fwb.13098

Naiman, R.J., Dudgeon, D., 2011. Global alteration of freshwaters: influences on human and environmental well-being. Ecol. Res. 26, 865–873.https://doi.org/10.1007/s11284-010-0693-3

The Nature Conservancy,, 2009. Indicators of Hydrologic Alteration Version 6.1 User's Manual. Available online (accessed on 20th February 2020, https://www.conservationgateway.org/ConservationPractices/Freshwater/EnvironmentalFlows/MethodsandTools/IndicatorsofHydrologicAlteration/Pages/IHA-Software-Download.aspx

Niedda, M., Pirastru, M., 2014. Field investigation and modelling of coupled stream discharge and shallow water-table dynamics in a small Mediterranean catchment (Sardinia). Hydrol. Process. 28 (21), 5423–5435. https://doi.org/10.1002/hyp.10016.

Nikolaidis, N.P., Demetropoulou, L., Froebrich, J., Jacobs, C., Gallart, F., Prat, N., Lo Porto, A., Campana, C., Papadoulakis, V., Skoulikidis, N., Davy, T., Bidoglio, G., Bouraoui, F., Kirkby, M., Tournoud, M.G., Polesello, S., Barberá, G.G., Cooper, D., Gomez, R., Sánchez-Montoya, M.M., Latron, J., De Girolamo, A.M., Perrin, J.L., 2013. Towards sustainable management of Mediterranean river basins: policy recommendations on management aspects of temporary streams. Water Policy 15, 830–849. https://doi.org/10.2166/wp.2013.158

Noguera Ropero, E., 2016. Temporalitat dels rius mediterranis: comunitat de peixos (Fish communities in Mediterranean temporary rivers, in Catalan). MSc thesis, Universitat de Barcelona, Catalonia, 69 p. Available at http://www.ub.edu/fem/docs/treballs/TFG_Eva_Noguera2016.pdf Accessed 29 July 19.

Obermann, M., Rosenwinkel, K. H.,Tournoud, M. G., 2009. Investigation of first flushes in a medium-sized mediterranean catchment. J. Hydrol. 373, 405-415. http://doi.org/10.1016/j.jhydrol.2009.04.038

Ollero, A., Ibisate, A., Gonzalo, L. E., Acín, V., Ballarín, D., Díaz, E., ... & Mora, D., 2011. The IHG index for hydromorphological quality assessment of rivers and streams: updated version. Limnetica, 30(2), 0255-262.

Olson, D.M., Dinerstein, E., Wikramanayake, E.D., Burgess, N.D., Powell, G.V.N., Underwood, E.C., D'amico, J.A., Itoua, I., Strand, H.E., Morrison, J.C., Loucks, C.J., Allnutt, T.F., Ricketts, T.H., Kura, Y., Lamoreux, J.F., Wettengel, W.W., Hedao, P., Kassem, K.R., 2001. Terrestrial ecoregions of the world: a new map of life on Earth. Bioscience 51, 933–938. https://doi.org/10.1641/0006-3568(2001)051[0933:TEOTWA]2.0.CO;2

Paisley, M.F., Trigg, D.J., Walley, W.J., 2014. Revision of the biological monitoring working party (BMWP) score system: derivation of present-only and abundance-related scores from field data. River Res. Appl. 30, 887–904. https://doi.org/10.1002/rra.2686

Pajunen, V., Luoto, M., Soininen, J., 2016. Climate is an important driver for stream diatom distributions. Global Ecol. Biogeogr. 25, 198–206. https://doi.org/10.1111/geb.12399

Papastergiadou, E., Manolaki, P., 2012. Developing an assessment system of RM-4 & RM-5 river types for Cyprus rivers. Final Report of Project TAY 84/2009. Natural Resources and Environment, Water Development Department, Ministry of Agriculture, Cyprus.

Parasiewicz, P., Rogers, J. N., Vezza, P., Gortázar, J., Seager, T., Pegg, M., ... & Comoglio, C., 2013. Applications of the MesoHABSIM simulation model. Ecohydraulics: an integrated approach, 109-124.

Pařil, P., Polášek, M., Loskotová, B., Straka, M., Crabot, J., & Datry, T., 2019. An unexpected source of invertebrate community recovery in intermittent streams from a humid continental climate. Freshwater Biology, 64(11), 1971-1983. https://doi.org/10.1111/fwb.13386

Pascual, U., Muradian, R., Brander, L., Gómez-Baggethun, E., Martín-López, B., Verma, M., ... & Farley, J., 2010. The economics of valuing ecosystem services and biodiversity. The economics of ecosystems and biodiversity: ecological and economic foundations, 183-256.

Pastor, A.V., Ludwig, F., Biemans, H., Hoff, H., Kabat, P., 2014. Accounting for environmental flow requirements in global water assessments. Hydrol. Earth Syst. Sci. 18. https://doi.org/10.5194/hess-18-5041-2014

Pastor, A. V, Palazzo, A., Havlik, P., Biemans, H., Wada, Y., Obersteiner, M., Kabat, P., Ludwig, F., 2019. The global nexus of food–trade–water sustaining environmental flows by 2050. Nat. Sustain. https://doi.org/10.1038/s41893-019-0287-1

Pawlowski, J., Kelly-Quinn, M., Altermatt, F., Apothéloz-Perret-Gentil, L., Beja, P., Boggero, A., et al., 2018. The future of biotic indices in the ecogenomic era: Integrating (e)DNA metabarcoding in biological assessment of aquatic ecosystems. Sci. Total Environ. 637, 1295–1310. https://doi.org/10.1016/j.scitotenv.2018.05.002

Pelte, T., Navarro, L., Stroffek, S., Dupre la Tour, J., Langon, M., Martinez, P.-J., Delhaye, H., Datry, T., 2012. Les cours d'eau intermittents, quelques repères pour une meilleure intégration aux plans de gestion. Note technique du SDAGE. Agence de l'Eau Rhône Méditerranée Corse, Lyon, France, 16 p.

Phillips, G., Kelly , M.,Teixeira, H.,Salas, F., Free, G., Leujak, W., et al., 2018. Best practice for establishing nutrient concentrations to support good ecological status, EUR 29329 EN, Publications Office of the European Union, Luxembourg, 2018, ISBN 978-92-79-92906-9. http://doi.org/10.2760/84425 JRC112667

Pilgrim, D. H., Chapman, T. G., Doran D. G., 1988. Problems of rainfall-runoff modelling in arid and semiarid regions, Hydrological Sciences Journal, 33:4, 379-400, DOI: 10.1080/02626668809491261

Poff, N., 1996. A hydrogeography of unregulated streams in the United States and an examination of scale-dependence in some hydrological descriptors. Freshwater biology, 36(1), 71-79. https://doi.org/10.1046/j.1365-2427.1996.00073.x

Poff, N.L., Allan, J.D., Bain, M.B., Karr, J.R., Prestegaard, K.L., Richter, B.D., …628 Stromberg, J.C., 1997 The Natural Flow Regime. BioScience, 47, 769–784. DOI: 10.2307/1313099

Poff, N. L., Richter, B. D., Arthington, A. H., Bunn, S. E., Naiman, R. J., Kendy, E., ... & Henriksen, J., 2010. The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards. Freshwater Biology, 55(1), 147-170. https://doi.org/10.1111/j.1365-2427.2009.02204.x

Poff, N.L., Zimmerman, J.K.H., 2010. Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. Freshw. Biol. 55, 194–205. https://doi.org/10.1111/j.1365-2427.2009.02272.x

Poznańska, M., Kakareko, T., Krzyżyński, M., Kobak, J., 2013. Effect of substratum drying on the survival and migrations of Ponto-Caspian and native gammarids (Crustacea: Amphipoda). Hydrobiologia 700, 47–59. https://doi.org/10.1007/s10750-012-1218-6

Prat, N., Gallart, F., Von Schiller, D., Polesello, S., García-Roger, E.M., Latron, J., Rieradevall, M., Llorens, P., Barberá, G.G., Brito, D., De Girolamo, A.M., Dieter, D., Lo Porto, A., Buffagni, A. Erba, S., Nikolaidis, N.P., Querner, E.P., Tournoud, M.G., Tzoraki, O., Skoulikidis, N., Gómez, R., Sánchez-Montoya, M.M., Tockner, K., Froebrich, J., 2014. The MIRAGE toolbox: an integrated assessment tool for temporary streams. River Res. Appl. 30, 1318–1334. https://doi.org/10.1002/rra.2757

Puckridge, J. T., Walker, K. F., & Costelloe, J. F., 2000. Hydrological persistence and the ecology of dryland rivers. Regulated Rivers: Research & Management: An International Journal Devoted to River Research and Management, 16(5), 385-402. https://doi.org/10.1002/1099-1646(200009/10)16:5<385::AID-RRR592>3.0.CO;2-W

Pumo, D., Caracciolo, D., Viola, F. and Noto, L. V., 2016. Climate change effects on the hydrological regime of small non-perennial river basins. Science of the Total Environment, 542, 76–92. https://doi.org/10.1016/j.scitotenv.2015.10.109

Querner, E.P., Froebrich, J., Gallart, F., Prat, N., Cazemier, M., Tzoraki, O., 2016. Simulating stream flow variability in temporary streams using a coupled groundwater-surface water model. Hydrological Sciences Journal, 1(1):146-161, doi: 10.1080/02626667.2014.983514.

Raven, P.J., Fox, P.J.A., Everard, M., Holmes, N.T.H., Dawson, F.H., 1997. River Habitat Survey: a new system for classifying rivers according to their habitat quality. In: Boon, P, Howell, D. (Eds.), Freshwater Quality: Defining the Indefinable? HMSO, Edinburgh, UK, pp. 215–234.

Reynolds, L. V., Shafroth, P. B., & Poff, N. L., 2015. Modeled intermittency risk for small streams in the Upper Colorado River Basin under climate change. Journal of Hydrology, 523, 768-780. https://doi.org/10.1016/j.jhydrol.2015.02.025

Richter, B.D., Baumgartner, J.V., Powell, J., and Braun, D.P., 1996. A method for assessing hydrologic alteration within ecosystems. Conservation Biology, 10(4), 1163-1174. https://doi.org/10.1046/j.1523-1739.1996.10041163.x

Rinaldi, M., Surian, N., Comiti, F., & Bussettini, M., 2013. A method for the assessment and analysis of the hydromorphological condition of Italian streams: The Morphological Quality Index (MQI). Geomorphology, 180, 96-108. https://doi.org/10.1016/j.geomorph.2012.09.009

Rinaldi, M., Surian, N., Comiti, F., & Bussettini, M., 2015. A methodological framework for hydromorphological assessment, analysis and monitoring (IDRAIM) aimed at promoting integrated river management. Geomorphology, 251, 122-136. https://doi.org/10.1016/j.geomorph.2015.05.010

Robinson, C.T., Tonolla, D., Imhof, B., Vukelic, R., Uehlinger, U., 2016. Flow intermittency, physico-chemistry and function of headwater streams in an Alpine glacial catchment. Aquat. Sci. 78, 327-341. http://doi.org/10.1007/s00027-015-0434-3

Robson, B. J., Chester, E. T., Mitchell, B. D., & Matthews, T. G., 2013. Disturbance and the role of refuges in mediterranean climate streams. Hydrobiologia, 719(1), 77-91. https://doi.org/10.1007/s10750-012-1371-y

Rodwell, J.S. (Ed.), 1995. British Plant Communities, Volume 4, Aquatic Communities, Swamps and Tall-herb Fens. Cambridge University Press, Cambridge, UK, 283 p.

Romaní, A.M., Amalfitano, S., Artigas, J., Fazi, S., Sabater, S., Timoner, X., et al., 2013. Microbial biofilm structure and organic matter use in mediterranean streams. Hydrobiologia. 719, 43-58. http://doi.org/10.1007/s10750-012-1302-y

Sabater, S., Timoner, X., Bornette, G., De Wildes, M., Stromberg, J.C., Stella, J.C., 2017. The biota of intermittent rivers and ephemeral streams: algae and vascular plants. In: Datry, T., Bonada, N., Boulton, A.J. (Eds.), Intermittent Rivers and Ephemeral Streams: Ecology and Management. Elsevier, Amsterdam, the Netherlands, pp. 189–216. https://doi.org/10.1016/B978-0-12-803835-2.00016-4

Sánchez-Montoya, M.M., Vidal-Abarca, M.R., Suarez, M.L., 2010. Comparing the sensitivity of diverse macroinvertebrate metrics to a multiple stressor gradient in Mediterranean streams and its influence on the assessment of ecological status. Ecol. Indic. 10, 896–904. https://doi.org/10.1016/j.ecolind.2010.01.008

Sánchez-Montoya, M.d.M., Arce, M.I., Vidal-Abarca, M.R., Suárez, M.L., Prat, N., Gómez, R., 2012. Establishing physico-chemical reference conditions in Mediterranean streams according to the European Water Framework Directive. Water Res 46, 2257-2269. http://doi.org/10.1016/j.watres.2012.01.042

Sánchez-Montoya, M.D.M., von Schiller, D., Ruhí, A., Pechar, G.S., Proia, L., Miñano, J., Vidal-Abarca, M.R., Suárez, M.L., Tockner, K., 2016. Responses of ground-dwelling arthropods to surface flow drying in channels and adjacent habitats along Mediterranean streams. Ecohydrology 9, 1376–1387. https://doi.org/10.1002/eco.1733

Sánchez-Montoya, M. M., Moleón, M., Sánchez-Zapata, J. A., & Tockner, K., 2016. Dry riverbeds: corridors for terrestrial vertebrates. Ecosphere, 7(10), e01508. https://doi.org/10.1002/ecs2.1508

Santos-Martín, F., Kelemen, E., García-Llorente, M., Jacobs, S., Oteros-Rozas, E., Barton, D. N., ... & Martín-López, B., 2017. 4.2. Socio-cultural valuation approaches. In: Burkhard, B. and J. Maes (eds.) *Mapping Ecosystem Services* (pp. 102-112). Pensoft Publishers Ltd, Sofia.

Sarremejane, R., Cañedo-Argüelles, M., Prat, N., Mykrä, H., Muotka, T., Bonada, N., 2017. Do metacommunities vary through time? Intermittent rivers as model systems. J. Biogeogr. 44, 2752–2763. https://doi.org/10.1111/jbi.13077

Sarremejane, R., Cid, N., Alp, M., Cañedo-Argüelles, M., Cordero-Rivera, A., Csabai, Z., Datry, T., Gutiérrez-Cánovas, C., Heino, J., Millán, A., Paillex, A., Paril, P., Polasek, M., Stubbington, R., Tierno de Figueroa, J.M., Usseglio-Polatera, P., Zamora-Muñoz, C., Bonada, N. (in prep.). A trait database to assess aquatic invertebrate dispersal potential.

Sarremejane, R., Stubbington, R., Dunbar, M.J., Westwood, C.G., England, J., 2019. Biological indices to characterize community responses to drying in streams with contrasting flow permanence regimes. Ecol. Indic. 107, 105620. https://doi.org/10.1016/j.ecolind.2019.105620

Sauquet, E., van Meerveld, I., Gallart, F., Sefton, C., Parry, S., Gauster, T., Laaha, G., Alves, M.H., Arnaud, P., Banasik, K., Beaufort, A., Bezdan, A., Datry, T., De Girolamo, A.M., Dörflinger, G., Elçi A., Engeland, K., Estrany, J., Fialho, A., Fortesa, J., Hakoun, V., Karagiozova T., Kohnova, S., Kriauciuniene, J., Morais, M., Ninov, P., Osuch, M., Reis, E., Rutkowska, A., Stubbington, R., Tzoraki, O., Żelazny, M., 2020. A catalogue of European intermittent rivers and ephemeral streams. Technical report SMIRES COST Action CA15113, 100 pp. doi: 10.5281/zenodo.3763419

Seaman, M., Watson, M., Avenant, M., King, J., Joubert, A., Barker, C., ... & Prucha, B., 2016. DRIFT-ARID: A method for assessing environmental water requirements (EWRs) for non-perennial rivers. Water SA, 42(3), 356-366. DOI: 10.4314/wsa.v42i3.01

Schneider, C., Laizé, C.L.R., Acreman, M.C. and Flörke, M., 2013 How will climate change modify river flow regimes in Europe? Hydrology and Earth System Sciences, 17, 325–339. https://doi.org/10.5194/hess-17-325-2013

Shumilova, O., Zak, D., Datry, T., von Schiller, D., Corti, R., Foulquier, A., et al., 2019. Simulating rewetting events in intermittent rivers and ephemeral streams: A global analysis of leached nutrients and organic matter. Glob. Change Biol. 25, 1591-1611. http://doi.org/10.1111/gcb.14537

Skoulikidis, N.T., Vardakas, L., Karaouzas, I., Economou, A.N., Dimitriou, E., Zogaris, S., 2011. Assessing water stress in Mediterranean lotic systems: insights from an artificially intermittent river in Greece. Aquat. Sci. 73, 581.doi:10.1577/TfW-089.1

Skoulikidis, N.T., Sabater, S., Datry, T., Morais, M.M., Buffagni, A., Dörflinger, G., Zogaris, S., del Mar Sánchez-Montoya, M., Bonada, N., Kalogianni, E., Rosado, J., Vardakas, L., De Girolamo, A. M., Tockner, K. 2017. Non-perennial Mediterranean rivers in Europe: status, pressures, and challenges for research and management. Sci. Total Environ., 577, 1–18, https://doi.org/10.1016/j.scitotenv.2016.10.147

Soininen, J., Jamoneau, A., Rosebery, J., Leboucher, T., Wang, J., Kokociński, M., Passy, S.I., 2019. Stream diatoms exhibit weak niche conservation along global environmental and climatic gradients. Ecography 42, 346–353. https://doi.org/10.1111/ecog.03828

Soria, M., Gutiérrez-Cánovas, C., Bonada, N., Acosta, R., Rodríguez-Lozano, P., Fortuño, F., Burgazzi, G., Vinyoles, D., Gallart, F., Latron, J., Llorens, P., Prat, N., Cid, N. (2020). Natural disturbances may produce misleading bioassessment results: identifying metrics to detect anthropogenic impacts in intermittent rivers. doi: 10.1111/1365-2664.13538

Soria, M., Leigh, C., Datry, T., Bini, L.M., Bonada, N., 2017. Biodiversity in perennial and intermittent rivers: a meta-analysis. Oikos 126, 1078–1089. https://doi.org/10.1111/oik.04118

Snelder, T. H., Datry, T., Lamouroux, N., Larned, S. T., Sauquet, E., Pella, H., & Catalogne, C., 2013. Regionalization of patterns of flow intermittence from gauging station records. 10.5194/hess-17-2685-2013

Snover, A. K., Hamlet, A. F., Lettenmaier D. P., 2003. Climate-change scenarios for water planning studies. Bulletin of the American Meteorological Society, 84(11), 1513–1518.

Stanley, E.H., Fisher, S.G., Jones, J.B., 2004. Effects of water loss on primary production: a landscape-scale model. Aquat. Sci. 66, 130–138. https://doi.org/10.1007/s00027-003-0646-9

Steward, A.L., Langhans, S.D., Corti, R., Datry, T., 2017. The biota of intermittent rivers and ephemeral streams: terrestrial and semi aquatic invertebrates. In: Datry, T., Bonada, N., Boulton, A.J. (Eds.), Intermittent Rivers and Ephemeral Streams: Ecology and Management. Amsterdam, the Netherlands, Elsevier, pp. 245–271 https://doi.org/10.1016/B978-0-12-803835-2.00008-5

Steward, A.L., Marshall, J.C., Sheldon, F., Harch, B., Choy, S., Bunn, S.E., Tockner, K., 2011. Terrestrial invertebrates of dry river beds are not simply subsets of riparian assemblages. Aquat. Sci. 73, 551–566. https://doi.org/10.1007/s00027-011-0217-4

Steward, A.L., Negus, P., Marshall, J.C., Clifford, S.E., Dent, C., 2018. Assessing the ecological health of rivers when they are dry. Ecol. Indic. 85, 537–547. https://doi.org/10.1016/j.ecolind.2017.10.053

Straka, M., Polášek, M., Syrovátka, V., Stubbington, R., Zahrádková, S., Němejcová, D., Šikulová, L. Řezníčková, P., Opatřilová, L., Datry, T., P. Pařil., 2019. Recognition of stream drying based on benthic macroinvertebrates: A new tool in Central Europe. Ecol. Indic. 106, 105486. https://doi.org/10.1016/j.ecolind.2019.105486

Stromberg, J.C., Merritt, D., 2016. Riparian plant guilds of ephemeral, intermittent and perennial rivers. Freshwater Biol. 61, 1259–1275. https://doi.org/10.1111/fwb.12686

Stubbington, R., 2012. The hyporheic zone as an invertebrate refuge: a review of variability in space, time, taxa and behaviour. Mar. Freshwater Res. 63, 293–311. https://doi.org/10.1071/MF11196

Stubbington, R., Datry, T., 2013. The macroinvertebrate seedbank promotes community persistence in temporary rivers across climate zones. Freshwater Biol. 58, 1202–1220. https://doi.org/10.1111/fwb.12121

Stubbington, R., Bogan, M.T., Bonada, N., Boulton, A.J., Datry, T., Leigh, C., Vander Vorste, R., 2017b. The biota of intermittent rivers and ephemeral streams: aquatic invertebrates. In: Datry, T., Bonada, N., Boulton, A.J. (Eds.), Intermittent Rivers and Ephemeral Streams: Ecology and Management. Elsevier, Amsterdam, the Netherlands, pp. 217–243. https://doi.org/10.1016/B978-0-12-803835-2.00007-3

Stubbington, R., Chadd, R., Cid, N., Csabai, Z., Miliša, M., Morais, M., Munné, A., Pařil, P., Pešić, V., Tziortzis, I., Verdonschot, R.C., 2018a. Biomonitoring of intermittent rivers and ephemeral streams in Europe: Current practice and priorities to enhance ecological status assessments. Sci. Total Environ., 618, 1096–1113. https://doi.org/10.116/j.scitotenv.2017.09.137

Stubbington, R., Dole-Olivier, M., Galassi, D.M.P., Hogan, J.P., Wood, P.J., 2016. Characterization of macroinvertebrate communities in the hyporheic zone of river ecosystems reflects the pump-sampling techniques used. PLoS One 11, 1–27. https://doi.org/10.1371/journal.pone.0164372

Stubbington, R., England, J., Acreman, M., Wood, P.J., Westwood, C., Boon, P., Mainstone, C., Macadam, C., Bates, A., House, A, Jorda-Capdevila, D., 2018b The Natural

Capital of Temporary Rivers: Characterising the value of dynamic aquatic-terrestrial habitats. Valuing Nature Natural Capital Synthesis Report VNP12. Available at: https://valuing-nature.net/TemporaryRiverNC Accessed 28 July 19.

Stubbington, R., England, J., Wood, P.J., Sefton, C.E., 2017a. Temporary streams in temperate zones: recognizing, monitoring and restoring transitional aquatic-terrestrial ecosystems. Wiley Interdiscip. Rev.: Water, 4, p.e1223. https://doi.org/10.1002/wat2.1223

Stubbington, R., Milner, V.S., Wood, P.J., 2019b. Flow intermittence in river networks: towards a global understanding of ecological diversity in aquatic–terrestrial ecosystems. Fundam. Appl. Limnol. 193, 1–19. https://doi.org/10.1127/fal/2019/1265

Stubbington, R., Paillex, A., England, J., Barthès, A., Bouchez, A., Rimet, F., Sánchez-Montoya, M.M., Westwood, C.G., Datry, T., 2019a. A comparison of biotic groups as dry-phase indicators of ecological quality in intermittent rivers and ephemeral streams. Ecol. Indic. 97, 165–174. https://doi.org/10.1016/j.ecolind.2018.09.061

Stubbington R., Acreman, M., Acuña, V., Boon, P.J., Boulton, A.J., England, J., Gilvear, D., Sykes, T., and Wood, P.J. (resubmitted). Ecosystem services of temporary streams differ between wet and dry phases in regions with contrasting climates and economies. *People and Nature*

Suding, K.N., Lavorel, S., Chapin, F.S., Cornelissen, J.H.C., Díaz, S., Garnier, E., Goldberg, D., Hooper, D.U., Jackson, S.T., Navas, M.L., 2008. Scaling environmental change through the community-level: A trait-based response-and-effect framework for plants. Global Change Biol. 14, 1125–1140. https://doi.org/10.1111/j.1365-2486.2008.01557.x

Theodoropoulos, C., Papadaki, C., Vardakas, L., Dimitriou, E., Kalogianni, E., Skoulikidis N., 2019. Conceptualization and pilot application of a model-based environmental flow assessment adapted for intermittent rivers. Aquatic Science, 81, 10 (2019). https://doi.org/10.1007/s00027-018-0605-0

Thoms, M. C., & Sheldon, F., 2000. Water resource development and hydrological change in a large dryland river: the Barwon–Darling River, Australia. Journal of Hydrology, 228(1-2), 10-21. https://doi.org/10.1016/S0022-1694(99)00191-2

Tornés, E., Ruhí A., 2013. Flow intermittency decreases nestedness and specialisation of diatom communities in Mediterranean rivers. Freshwater. Biol. 58, 2555–2566. https://doi.org/10.1111/fwb.12232

Trancoso R., Braunschweig F., Chambel- Leitao P., Neves R., 2009. An advanced modeling tool for simulating complex river systems. Science of The Total Environment, 407(8):3004-16, doi: 10.1016/j.scitotenv.2009.01.015

Tzoraki O., Nikolaidis N, Trancoso R., Neves R., Braunschweig F., 2009. Modeling of in-stream biogeochemical processes of temporary rivers. Hydrological processes, 23(2), 272-283.

Uys, M. C., & O'keeffe, J. H., 1997. Simple words and fuzzy zones: early directions for temporary river research in South Africa. Environmental Management, 21(4), 517-531. DOI: 10.1007/s002679900047

Van de Bund, W. (Ed.), 2009. Water Framework Directive intercalibration technical report. Part 1: Rivers. JRC Scientific and Technical Reports EUR 23838 EN/1 – 2009. Office for Official Publications of the European Communities, Luxembourg, https://doi.org/10.2788/23384 Accessed 27 July 19.

Vander Vorste, R., Malard, F., & Datry, T., 2016. Is drift the primary process promoting the resilience of river invertebrate communities? A manipulative field experiment in an intermittent alluvial river. Freshwater Biology, 61(8), 1276-1292. https://doi.org/10.1111/fwb.12658

Vardakas, L., 2017. Spatial and temporal patterns of fish assemblages in a Mediterranean intermittent river. PhD thesis, University of the Aegean, Greece, 233 p.

Vasselon, V., Bouchez, A., Rimet, F., Jacquet, S., Trobajo, R., Corniquel, M., Tapolczai, K., Domaizon, I., 2018. Avoiding quantification bias in metabarcoding: Application of a cell biovolume correction factor in diatom molecular biomonitoring. Methods Ecol. Evol. 9, 1060–1069. https://doi.org/10.1111/2041-210X.12960

Vassoney E., Mammoliti Mochet A., Rocco R., Maddalena R., Vezza P., Comoglio C., 2019. Integrating meso-scale habitat modelling in the multicriteria analysis (MCA) process for the assessment of hydropower sustainability. Water. 11(4),640. https://doi.org/10.3390/w11040640.

Vericat, D., Batalla R.J., 2010. Sediment transport from continuous monitoring in a perennial Mediterranean stream. Catena. 82, 77-86. https://doi.org/10.1016/j.catena.2010.05.003

Vezza P., Parasiewicz P., Spairani M., Comoglio C., 2014. Habitat modelling in high gradient streams: the meso-scale approach and application. Ecological Applications. 24(4):844-861. http://dx.doi.org/10.1890/11-2066.1

Viola, F., Pumo D., Noto L. V., 2014. EHSM: A conceptual ecohydrological model for daily streamflow simulation, Hydrol. Processes, 28(9), 3361–3372

Von Schiller, D., Bernal, S., Dahm, C.N., Martí, E., 2017a. Nutrient and Organic Matter Dynamics in Intermittent Rivers and Ephemeral Streams. In Intermittent Rivers and Ephemeral Streams, eds. T. Datry, N. Bonada, A. Boulton, 135-160. Academic Press.

Von Schiller, D., Acuña, V., Aristi, I., Arroita, M., Basaguren, A., Bellin, A., et al., 2017b. River ecosystem processes: A synthesis of approaches, criteria of use and sensitivity to environmental stressors. Sci. Tot. Environ. 596, 465-480. doi.org/10. http://doi.org/1016/j.scitotenv.2017.04.081

Vörösmarty, C.J., McIntyre, P.B., Gessner, M.O., Dudgeon, D., Prusevich, A., Green, P., Glidden, S., Bunn, S.E., Sullivan, C.A., Liermann, C.R., Davies, P.M., 2010. Global threats to human water security and river biodiversity . Nature 468, 334. https://doi.org/doi:10.1038/Nature09549

Vörösmarty, C.J., Hoekstra, A.Y., Bunn, S.E., Conway, D., Gupta, J., 2015. Fresh water goes global. Science (80-.). 349, 478–479. DOI: 10.1126/science.aac6009

Walling DE, Fang D. 2003. Recent trends in the suspended sediment loads of the world's rivers. Global and Planetary Change 39 : 111–126. https://doi.org/10.1016/S0921-8181(03)00020-1

Warfe, D.M., Pettit, N.E., Davies, P.M., Pusey, B.J., Hamilton, S.K., Kennard, M.J., Townsend, S.A., Bayliss, P., Ward, D.P., Douglas, M.M., 2011. The 'wet–dry' in the wet–dry tropics drives river ecosystem structure and processes in northern Australia. Freshw. Biol. 56, 2169–2195. https://doi.org/10.1111/j.1365-2427.2011.02660.x

Weigand, H., Beermann, A.J., Čiampor, F., Costa, F.O., Csabai, Z., Duarte, S., Geiger, M.F, Grabowski, M., Rimet, F., Rulik, B., Strand, M., Szucsich, N., Weigand, A.M., Willassen, E., Wyler, S.A., Bouchez, A., Borja, A., Čiamporová-Zaťovičová, Z., Ferreira, S., Dijkstra, K.D., Eisendle, U., Freyhof, J., Gadawski, P. Graf, W., Haegerbaeumer, A., van der Hoorn, B.B., Japoshvili, B. Keresztes, L. Keskin, E., Leese, F., Macher, J., Mamos, T., Paz, G., Pešić, V., Pfannkuchen, D.M., Pfannkuchen, M.A., Price, B.W., Rinkevich, B., Teixeira, M.A.L., Várbíró, G., Ekrem, T., 2019. DNA barcode reference libraries for the monitoring of aquatic biota in Europe: Gap-analysis and recommendations for future work. Sci. Total Environ. 678, 499–524. https://doi.org/10.1016/j.scitotenv.2019.04.247

Westwood. C.G., England J., Johns, T., Stubbington R., in prep. A new method to assess intermittent rivers, using macrophytes.

Westwood, C.G., Teeuw, R.M., Wade, P.M., Holmes, N.T.H., Guyard, P., 2006a. Influences of environmental conditions on macrophyte communities in drought-affected headwater streams. River Res. Appl. 22, 703–726. https://doi.org/10.1002/rra.934

Westwood, C.G., Teeuw, R.M., Wade, P.M., Holmes, N.T.H., 2006b. Prediction of macrophyte communities in drought-affected groundwater-fed headwater streams. Hydrol. Process. 20, 127–145. https://doi.org/10.1002/hyp.5907

Wheater, H., Sorooshian, S., Sharma, K.D., 2007. Hydrological Modelling in Arid and Semi-Arid Areas. Cambridge University Press. https://doi.org/10.1017/

White, J., Armitage, P., Bass, J., Chadd, R., Hill, M., Mathers, K., Little, S., Wood, P.J., 2019. How freshwater biomonitoring tools vary sub-seasonally reflects temporary river flow regimes: Biomonitoring tool responses to temporary river flow regimes. River Res. Appl. https://doi.org/10.1002/rra.3501

White, J.C., House, A., Punchard, N., Hannah, D.M., Wilding, N.A., Wood, P.J., 2018. Macroinvertebrate community responses to hydrological controls and groundwater abstraction effects across intermittent and perennial headwater streams. Sci. Total Environ. 610, 1514–1526. https://doi.org/10.1016/j.scitotenv.2017.06.081

Wilding, N.A., White, J.C., Chadd, R.P., House, A., Wood, P.J., 2018. The influence of flow permanence and drying pattern on macroinvertebrate biomonitoring tools used in the assessment of riverine ecosystems. Ecol. Indic. 85, 548–555. https://doi.org/10.1016/j.ecolind.2017.10.059

Wilkes, M.A., Edwards, F., Jones, J.I., Murphy, J.F., England, J., Friberg, N., Hering, D., Poff, N.L., Usseglio-Polatera, P., Verberk, W., Webb, J., Brown, L.E. (in prep.) “Big” trait-based ecology: Assessing functional trait correlations, phylogenetic constraints and spatial instability with open data.

Wohl, E., Bledsoe, B. P., Jacobson, R. B., Poff, N. L., Rathburn, S. L., Walters, D. M., & Wilcox, A. C., 2015. The natural sediment regime in rivers: broadening the foundation for ecosystem management. *BioScience*, *65*(4), 358-371.https://doi.org/10.1093/biosci/biv002

Wyżga, B., Zawiejska, J., Radecki-Pawlik, A., & Amirowicz, A., 2010. A method for the assessment of hydromorphological river quality and its application to the Czarny Dunajec, Polish Carpathians. Cultural landscapes of river valleys. Agricultural University in Kraków, Kraków, 145-164.

Wyżga, B., Zawiejska, J., Radecki-Pawlik, A., & Hajdukiewicz, H., 2012. Environmental change, hydromorphological reference conditions and the restoration of Polish Carpathian rivers. Earth Surface Processes and Landforms, 37(11), 1213-1226. https://doi.org/10.1002/esp.3273

Ye, W, Bates, B.C., Viney, N.R., Sivapalan, M., Jakeman, A.J., 1997. Performance of conceptual rainfall-runoff models in low-yielding ephemeral catchments. Water Resources Research 33: 153–166.

Ye, W., Jakeman, A. J., Young, P. C., 1998, Identification of improved rainfall-runoff models for an ephemeral low-yielding Australian catchment, Environ. Modell. Softw., 13(1), 59–74.

Ylla, I., Sanpera-Calbet, Vázquez, E., Romaní, A.M., Muñoz, I., Butturini, A., Sabater, S., 2010. Organic matter availability during pre- and post-drought periods in a Mediterranean stream. Hydrobiologia 657, 217–232. http://doi.org/10.1007/s10750-010-0193-z

Yu, S., Bond, N. R., Bunn, S. E., Xu, Z., Kennard, M. J., 2018. Quantifying spatial and temporal patterns of flow intermittency using spatially contiguous runoff data. Journal of Hydrology, 559(1), 861–872.

Zimmer, M. A., Kaiser, K. E., Blaszczak, J. R., Zipper, S. C., Hammond, J. C., Fritz, K. M., ... & Kampf, S., 2020 Zero or not? Causes and consequences of zero-flow stream gage readings. Wiley Interdisciplinary Reviews: Water, e1436. https://doi.org/10.1002/wat2.1436

Zogaris, S., Tachos, V., Economou, A., Chatzinikolaou, Y., Koutsikos, N., Schmutz S., 2018. A model-based fish bioassessment index for Eastern Mediterranean rivers: Application in a biogeographically diverse area. Sci. Total Env. 622, 676–689. https://doi.org/10.1016/j.scitotenv.2017.11.293

Zoppini, A., Ademollo, N., Amalfitano, S., Casella, P., Patrolecco, L., Polesello, S., 2014. Organic priority substances and microbial processes in river sediments subject to contrasting hydrological conditions. Sci. Tot. Environ. 484, 74-83. http://doi.org/10.1016/j.scitotenv.2014.03.019

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