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Improved modelling of the freshwater provisioning ecosystem service in water scarce river basins

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Highlights

- The freshwater provisioning ecosystem service is influenced by water management
- Water resources management models capture this better than ecosystem service tools
- We link hydrologic, water allocation and water quality models to assess the service
- They fit temporal and spatial details of ecological processes and water management

Abstract

Freshwater provisioning by the landscape contributes to human well-being through water use for drinking, irrigation and other purposes. The assessment of this ecosystem service involves the quantification of water resources and the valuation of water use benefits. Models especially designed to assess ecosystem services can be used. However, they have limitations in representing the delivery of the service in water scarce river basins where water management and the temporal variability of water resource and its use are key aspects to consider. Integrating water resources management tools represents a good alternative to ecosystem services models in these river basins. We propose a modelling framework that links a rainfall-runoff model and a water allocation model which allow accounting for the specific requirements of water scarce river basins. Moreover, we develop a water tracer which rebounds the value of the service from beneficiaries to water sources, allowing the spatial mapping of the service.

Keywords: ecosystem services; freshwater provisioning; water resources management; integrated modelling; water scarcity; AQUATOOL

Software availability: Downloads of the software used in the presented analysis are available in http://www.upv.es/aquatool/en/software_en.html

Introduction

The importance that the services provided by ecosystems (ecosystem services, ES) have for human well-being has gained broad recognition in the last decade. Lately, ES have been incorporated into the political and scientific international agenda as a way to support environmental protection and the efficient use of scarce resources. Outstanding examples are the Mapping and Assessment of Ecosystems and their Services (Maes et al., 2016) that assists EU member states in mapping and assessing the state of their ES with the aim of informing the development and implementation of related policies; the Natural Capital Project (Natural Capital Project, 2016), which proposes tools and approaches to account for nature’s contributions to society that are useful for decision makers; and the Intergovernmental Platform on Biodiversity and Ecosystem Services (Díaz et al., 2015), which assesses the state of biodiversity and of the ES it provides to society in response to requests from decision makers. All these big initiatives point out science-policy interaction as the way to apply the ES approach in practice. It is also in the background of these initiatives the need for bringing ES assessment to the operational level, in which planning and management of natural resources take place, in order to make the most of the ES approach and effectively advance to a more sustainable decision making. To do so, suitable tools to analyse the impact of management actions on ES are necessary (Connor et al., 2015).

In the case of water resources, the management scale is the river basin as established by the European Water Framework Directive (European Parliament and Council, 2000) and in line with the Integrated Water Resources Management paradigm (Global Water Partnership, 2000). Even though water is essential for most ecosystem processes that rely on water abundance, temporal and spatial distribution, there are only two types of ES that are related to its management. Aquatic ES account for the benefits provided by freshwater ecosystems such as water purification (Keeler et al., 2012; La Notte et al., 2012; Liquete et al., 2011; Terrado et al., 2016) and habitat for fish (Liquete et al., 2016; Sample et al., 2016). On the other hand, hydrologic ES describe the benefits to people derived from the relationship between terrestrial ecosystems and freshwater quantity and quality (Brauman, 2015); some examples are freshwater provision (Boithias et al., 2014; Dennedy-Frank et al., 2016; Guo et al., 2000; Karabulut et al., 2016; Terrado et al., 2014), flood mitigation (Fu et al., 2013; Watson et al., 2016) and pollution abatement (Bogdan et al., 2016; Fu et al., 2012).

Unlike aquatic ES, which are clearly related to water management, the relationship between hydrologic ES and water management is not straightforward. The biophysical processes that underpin them take place in the landscape and, thus, they are affected by landscape management in first place (Guswa et al., 2014). While this is true, the anthropocentric perspective of ES only accounts for their value as far as they provide direct or indirect benefits to people. This means that the water yielded by a landscape or the pollutants retained by its vegetation cannot be accounted for as ES if they are not beneficial for downstream humans. The use of water occurs in water bodies (i.e rivers, lakes and aquifers) whose natural flow and volume patterns are modified by hydraulic infrastructures and water management practices (Richter and Thomas, 2007). Hence, eventually, the economic value of hydrologic ES is influenced by water management. Although the extent of water management impacts in some river basins is not significant, it is very pronounced in arid and semi-arid river basins which suffer from endemic water scarcity (Grafton et al., 2013; Richter and Thomas, 2007). For this reason, the assessment of hydrologic ES in this kind of river basins should take into account the influence of water management when the objective is providing reliable and accurate information for decision making.

Bearing the above in mind, the selection of the model to assess hydrologic ES in water scarce river basins should be thorough. Simulation models especially designed for ES assessment, or ES tools, integrate ecological and economic aspects for several ES considering their spatial variability (Bagstad et al., 2013a). They allow analysing tradeoffs between ES under different scenarios and are attainable for non-experts (Terrado et al., 2014). An extensive review of ES tools can be found in (Bagstad et al., 2013a). The Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) (Tallis et al., 2013) is likely the most widely known ES tool. It is a spatially explicit model to estimate levels of different ES benefits in a static timeframe, usually an average year (Terrado et al., 2016). InVEST includes freshwater provisioning, sediment retention, and water purification as hydrologic ES. It accounts for the processes taking place in the landscape considering simplified hydrological relationships whose main input are land use-land cover maps linked to biophysical parameters such as roots depth and retention capacity of vegetation. The instream processes are also simplified and limited to the conveyance of water to its use location, without regarding the influence of water infrastructures and their operation.

Another well-known ES tool is the web-based Artificial Intelligence for Ecosystem Services (ARIES) (Villa et al., 2014). It applies a probabilistic Bayesian network approach which uses a library of models and spatial data to quantify ES flows and uncertainty when little data is available (Bagstad et al., 2013b), but it also allows employing biophysical relationships when enough data is accessible (Vigerstol and Aukema, 2011). The hydrologic ES addressed by ARIES are flood regulation, nutrient regulation, sediment regulation, and water supply. It works with a time step ranging from hours to years, and does not value the ES in economic units (Villa et al., 2014). Even though this ES tool is flexible to introduce instream processes, it lacks the capabilities to faithfully represent water management influence on the delivery of hydrologic ES. Moreover, the model complexity can hinder the understanding of the modelled processes and the results for decision makers and stakeholders (Vigerstol and Aukema, 2011).

Both InVEST and ARIES, and presumably the remaining ES tools, present serious drawbacks to be used for the assessment of hydrologic ES in water scarce regions in which natural river processes are affected by the intense exploitation of water resources and changing management rules. In this context, the models traditionally used for Integrated Water Resources Management (IWRM) are a good alternative to ES tools. The integrative approach of these models aim at realistically representing hydrological processes and water management effects on water availability, water quality and derived variables (Davies and Simonovic, 2011) with appropriate spatial and temporal resolution. Some examples are SWAT (Arnold et al., 1998) and HBV (Bergström, 1995) as rainfall-runoff models; SIMGES (Andreu et al., 1996) and WEAP (Yates et al., 2005) as water allocation models; GESCAL (Paredes-Arquiola et al., 2010) and QUAL2 (Chapra et al., 2005) as water quality models; and CAUDECO (Paredes-Arquiola et al., 2014b) and TSLIB (Milhous, 1990) as habitat suitability models. They have broad scientific recognition and are already in use in many water scarce river basins to support decision making (Vigerstol and Aukema, 2011). This makes them easy to adopt for ES assessments, despite that their higher complexity makes them more difficult to parameterise than most ES tools. Consequently, potential gains in accuracy should be balanced with the increase of complexity (Bagstad et al., 2013a) when it comes to applying IWRM tools for ES assessment.

This paper focuses on the assessment of the Freshwater Provisioning hydrologic ES (FPS). Brauman et al. (2007) define it as the natural process that modifies the quantity of water for extractive (e.g. drinking, irrigation and industrial uses) and on site purposes (e.g hydropower generation, water recreation and transport). The main aim of the study is proposing a modelling framework composed of IWRM models to assess the FPS with detailed consideration of water resources management impacts. The paper describes the linkage and adaptation of a rainfall-runoff model, a water allocation model and a water quality model to obtain the spatial distribution of the FPS in biophysical and economic units. To the best knowledge of the authors, a similar modelling approach has not been presented previously. The methodology is illustrated in the Tormes River Basin (TRB) in Spain, which has a predominant semi-arid climate, for two scenarios that introduce changes in the landscape and in water management with respect to the business as usual. Results demonstrate the influence of water management on the delivery of the service, which justifies the convenience of using IWRM models to make up for the limitations of ES tools in water scarce river basins.

Material and Methods

Modelling framework

The FPS is provided by the landscape where rainfall-runoff processes take place. Terrestrial ecosystems partly determine these processes with their influence on landscape features such as water retention capacity of soils, percolation or slope. Each part of the catchment has a different capacity to generate runoff in its diverse components (surface and groundwater water resources). As water reaches rivers, lakes and aquifers, it can be withdrawn by diverse water users that obtain a benefit from it; i.e. urban, agricultural, industrial and water-related recreational uses. Therefore, any tool used to conduct the assessment of the FPS should consider all these aspects. The proposed modelling framework (Figure 1) comprises a rainfall-runoff model (RRM) that represents the production of water resources; a water allocation model (WAM) which reproduces the use of water by the different beneficiaries of the service; economic functions (demand curves) that translate the use of water into economic benefits; and a water quality model that is used as a water resources tracer to assign the economic value of the service to the part of the catchment producing it (spatial mapping).

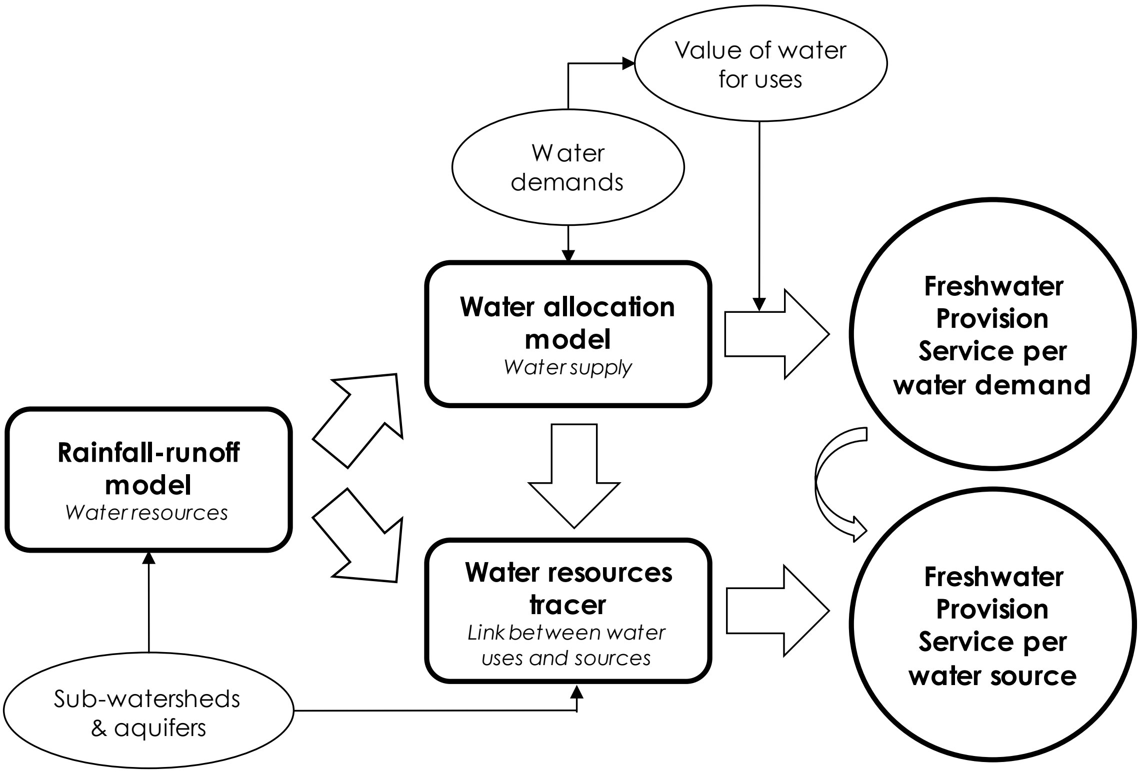

Figure 1. Modelling framework for the assessment of the FPS.

In the first place, meteorological data and hydrologic features are used to run the RRM, which provides runoff time series for all the water sources in the basin (i.e. sub-watersheds and aquifers). This requires the RRM to be spatially distributed or semi-distributed and to explicitly consider surface and groundwater components. For the purpose of analysing the impact of land use change scenarios on the FPS, it is advisable to use a physically based model (or at least a conceptual model) that allows translating landscape changes into parameters changes in a straightforward way. Furthermore, the spatial resolution of the model should be defined in agreement with the purpose of the assessment. Regarding the time step, since the purpose of the RRM here is not the obtaining of hydrographs but the assessment of available water resources, the month is regarded as convenient in terms of the representation of the seasonal variability of flows. The monthly step is also suitable to analyse most water management problems (Dyck, 1990). The WAM uses the RRM results and simulates the water flows along the regulated river system, considering the infrastructures and water management influence. The relevant outcomes for the presented framework are the time series of water supplied to each water use. The selection of the WAM depends on the data availability and the purpose of the study, but at least it should account for surface and groundwater interaction and abstraction, and be able to represent common water management strategies such as water supply priorities and operation rules.

Once the water resources are allocated, economic functions are used to assign a value to the use of water. According to Momblanch et al. (2016), production-based valuation methods should be used when the valued ES is a factor of production for a good or service traded on the market, while the aggregated willingness-to-pay is applied to establish the economic value of services that are goods whose market price does not include the impact of use on their availability for other users and the environment. In line with this, the marginal residual value of water for production is used to define the economic value of water for uses like agriculture and industry, whereas the aggregated willingness-to-pay is applied to establish the economic value of water for urban supply, recreation, and other final water uses (de Groot et al., 2002; Pulido-Velazquez et al., 2008). Commonly, hydro-economic models make use of the so-called water demand curves for the different water uses to capture all this information (Momblanch et al., 2016). Demand curves relate the volume (usually annual) of water supplied (Mm^3/year) to its unitary value (€/m^3); some examples can be found in Pulido-Velazquez et al. (2006). The gross benefit of certain water use is calculated as the integral under the water demand curve as shown in Figure 2.

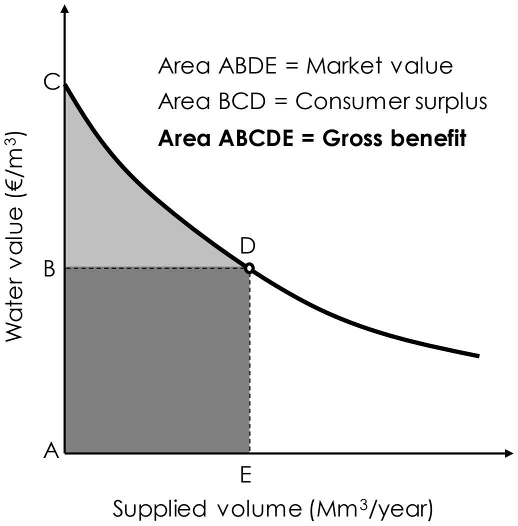

Figure 2. Obtaining of the gross benefit from a water demand curve.

Since the water supplies provided by the WAM have a monthly step, they are yearly accumulated to be compatible with the demand curves. The annual benefit resulting from the demand curves for each water use is then temporally distributed according to the monthly water supply. The total monthly benefit provided by the FPS in the whole river basin is calculated as the sum of the monthly benefits of all water uses. These results are helpful when analysing different water management strategies.

In order to evaluate catchment management actions, it is relevant to know the contribution of each water source to the global FPS benefit. In a non-regulated river basin, the best option would be sharing the FPS benefit as per the fraction of total water resources that each water source generates. However, the existence of infrastructures for storage and conveyance of water strongly affects the natural flow patterns and the proportional sharing of the benefits may not be realistic. For the spatial mapping of the service in water scarce river basins, the modelling framework accumulates the ES benefit per water source according to the fraction of the water supply that they provide to each demand. The relationship between the watershed or aquifer producing the water resource and the final water use which gives an economic value to the water supplied is not easily obtainable. As water is routed along the river network, reservoirs and canals by the WAM, it mixes and it is not possible to trace its origin in the landscape. The proposed FPS modelling framework makes use of a water tracer (see Figure 3) based on the iterative execution of mass balance simulations, considering the movement of water along the river system resulting from the WAM. To do so, a fictitious conservative pollutant (C) that is only affected by the convection driven by the water movement is defined using a mechanistic water quality model.

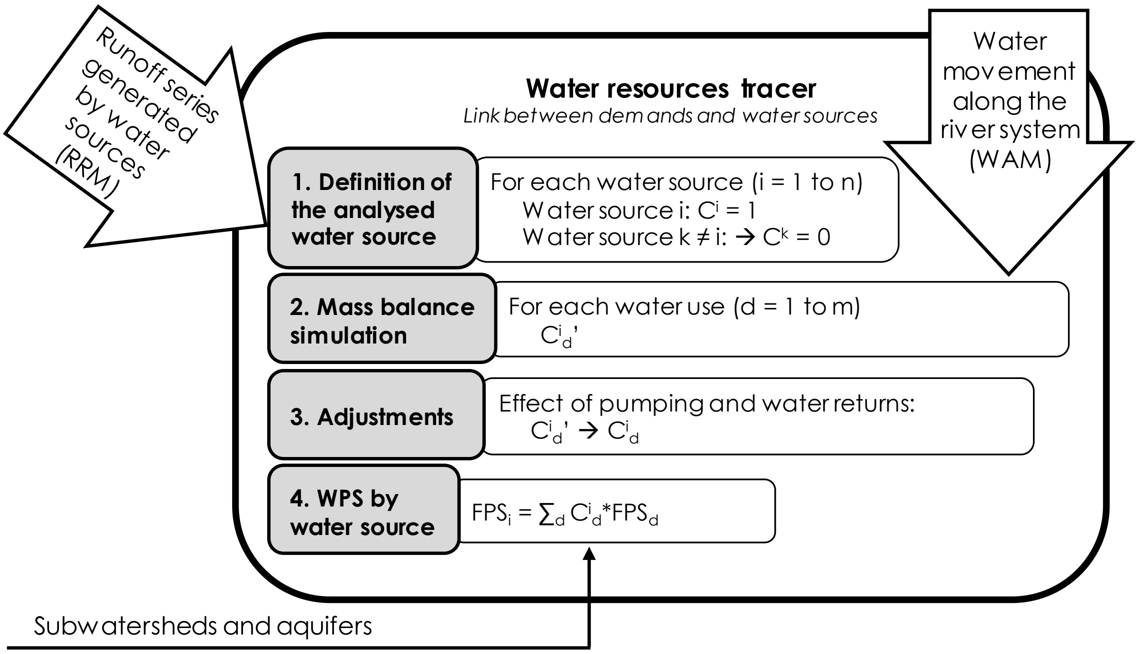

Figure 3. Water tracer diagram for the mapping of the FPS.

It is necessary to run one simulation per water source. In each simulation, the concentration of the fictitious pollutant equals to 0 in the water generated by all sources (C^k), except for the water source analysed in that specific execution of the tracer (i) for which the concentration equals to 1 (C^i). Given that the pollutant is conservative, its concentration only varies due to dilution in water with a different pollutant concentration. In this case, concentration changes when water from the analysed source is mixed with water coming from other sources. Therefore, the concentration of the fictitious pollutant in the water withdrawn by a water use (C^i_d') is equivalent to the fraction of the water supply to this water use originated in the analysed water source. This value should be recalculated for uses receiving pumped water since it does not get mixed with other water sources and its concentration remains constant, as opposed to groundwater runoff which propagates along the river system. In the case that water returns from water uses exist, part of the water resources generated by the sources upstream the use producing the return can be used more than once. Hence, it is necessary to conduct one additional simulation for each water return assigning it a concentration equal to 1. Knowing the proportion of the water returned that is used by the downstream uses, it is possible to adjust the fraction of water supplied by the upstream water sources to consider its indirect reuse. With this procedure, the water tracer provides $m \cdot n$ time series of C^i_d that represent the fraction of water supplied to each water use from each water source along time. The FPS per water source (FPS_i) is calculated as the sum of the FPS benefit for each water use (FPS_d) times the proportion of water that it receives from the analysed water source (C^i_d). A final aspect to highlight is the influence of the initial concentration of the fictitious pollutant in reservoirs on the results of the water tracer. Therefore, a warm-up period has to be considered in order to ensure that the results obtained are not biased by the initial concentration values assumed.

Study area: Tormes River Basin

The TRB belongs to the Duero River Basin District in Spain (see Figure 4). It covers an area of 9,568km^2 with an average precipitation of 529.9mm/year and a potential evapotranspiration of 826.28mm/year, resulting in a mean annual total runoff of 1,678.2Mm^3. It has a predominant semi-arid climate with Mediterranean and Continental influence. The TRB spans from the mountainous region of Sierra de Gredos and flows north-west until the convergence with the Duero River, just downstream La Almendra reservoir. It counts with large Natura 2000 sites at the heading and at the lower part of the basin.

The main water uses in the TRB are agriculture with a water demand of 319.5Mm^3/year, urban demands that amount to 38.9Mm^3/year, and hydropower uses that are mostly run-of-river stations and, hence, do not determine water management. The total population in the TRB is around 280,000 inhabitants of which more than 160,000 live in the city of Salamanca. Even though the basin holds several reservoirs, only Santa Teresa performs inter-annual regulation since La Almendra reservoir only serves downstream uses, which are outside the TRB. The conceptualisation of the basin considered in this application is a simplification of the real system. This is because the purpose of the application case is not getting insight of the real behaviour of the TRB, but exemplifying the type of analysis that the modelling framework allows in a simple and clear way. The simplified TRB only contains the urban demand of Salamanca with the highest supply priority, the irrigation uses grouped in three areas with equal supply priority, Santa Teresa reservoir, and the inflows generated by all sub-watersheds grouped into four (see Figure 4 and Figure 5).

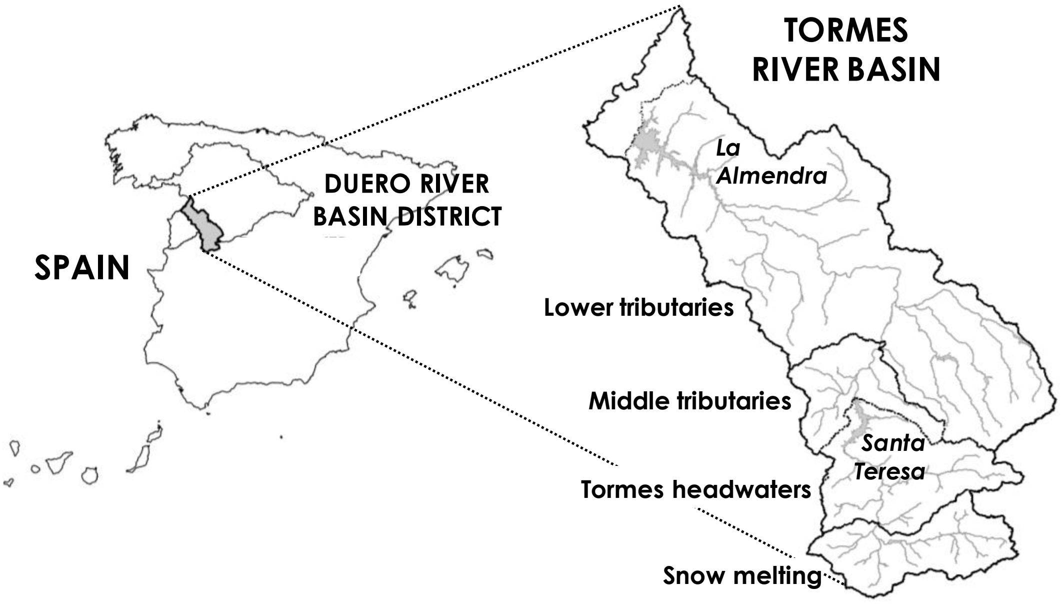

Figure 4. Location of the TRB, main reservoirs and sub-watersheds.

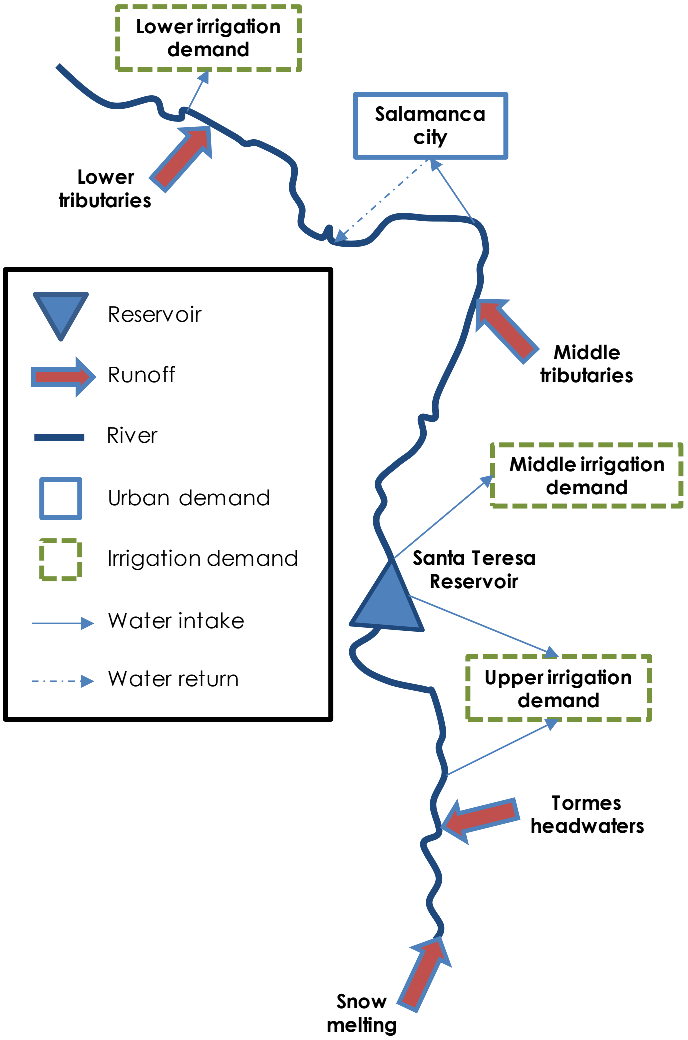

Figure 5. Diagram of the simplified TRB.

The analysis is performed for a period of 51 years with representative conditions of the system. For that purpose, we select the historical period comprising the hydrological years 1955 to 2006 (from October 1955 to September 2007), which cover a four-year dry episode from 1979 to 1983. For the analysed period, high resolution daily gridded datasets of climatic data are available (Herrera et al., 2012), as well as maps of soil characteristics to be used as inputs of the RRM. On the other hand, the WAM needs mean monthly data about water demands; reservoirs capacity, bathymetry and evaporation rates; the capacities of transport networks; etc. These data are available in the databases of the Duero River Basin Agency. Besides, the runoff flows entering the TRB from the basin headings and the tributaries are provided by the RRM. Finally, the demand curves are estimated by the Spanish Water Directorate in a specific report (Ministerio de Agricultura Alimentación y Medio Ambiente, 2012) which provided one demand curve for all urban uses in the Duero River Basin District, and one demand curve for the agricultural uses in the TRB. The demand curve for the Salamanca city is derived from the former, while the demand curves for the three irrigation areas are obtained from the latter (Figure 6).

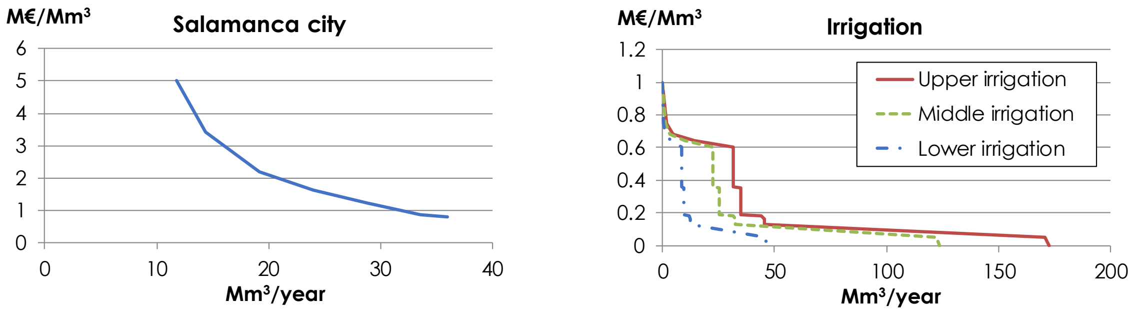

Figure 6. Demand curves adapted for the water uses in the simplified TRB.

IWRM tools

Many different IWRM models can be used to implement the presented modelling approach as far as they comply with the recommendations previously commented in this section. In our study, we assess the FPS in the TRB using models included in the Decision Support System environment AQUATOOL (Andreu et al., 1996) for water resources planning and management. It is a geo-referenced database system which provides a common interface, data and results management tools for different modules directed to analyse the key aspects of river basins and water resources systems. The software EVALHID (Paredes-Arquiola et al., 2014a) and SIMGES (Andreu et al., 1996) are used as RRM and WAM, respectively. The water tracer makes use of the water quality model GESCAL (Paredes-Arquiola et al., 2010).

For the setup of the modelling framework in the TRB, the WAM is manually calibrated using the observed and simulated volumes stored in Santa Teresa reservoir (see Figure 7), together with the flows just upstream La Almendra reservoir, for the period 1996-2006. It can be considered that the main infrastructures, water demands and management rules remain constant during this period. The calibration of the WAM is previous to the RRM and, thus, the model is fed with gauged inflows restored to the natural flow regime.

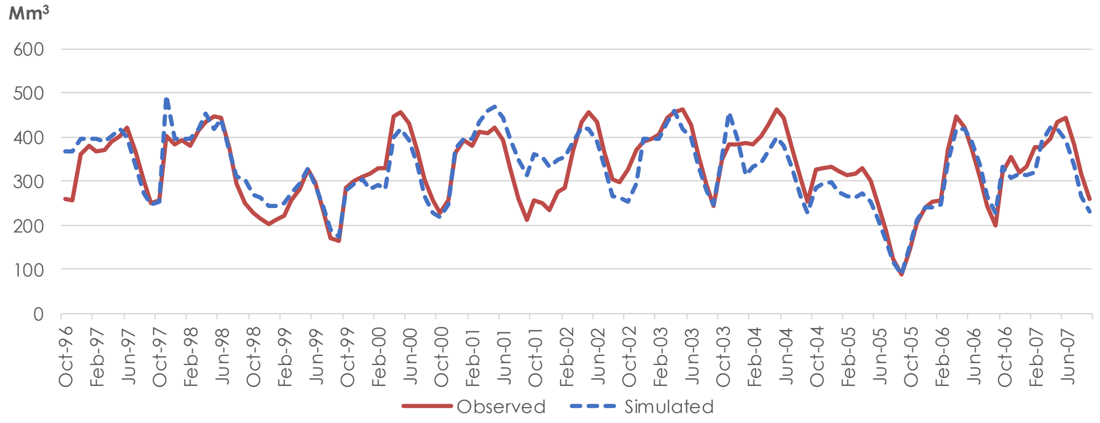

Figure 7. Comparison of observed and simulated volumes stored in Santa Teresa reservoir.

The RRM is built with EVALHID considering the conceptual model HBV (Bergström, 1995). Each sub-watershed in the TRB is calibrated using observed flows in the river for the period 1996-2006. Nevertheless, the flows generated by the RRM are in natural regime and they are not comparable with the gauged flows. Therefore, the RRM results are introduced as inputs to the calibrated WAM that affects them with the management conditions of the system, making possible the comparability of simulated and observed flows (see Figure 8). An automatic calibration process is performed using the Shuffled Complex Evolution Algorithm, SCEUA (Duan et al., 1994) based on the average of Nash-Sutcliffe, log Nash-Sutcliffe, Pearson's coefficient, and percent bias as target function.

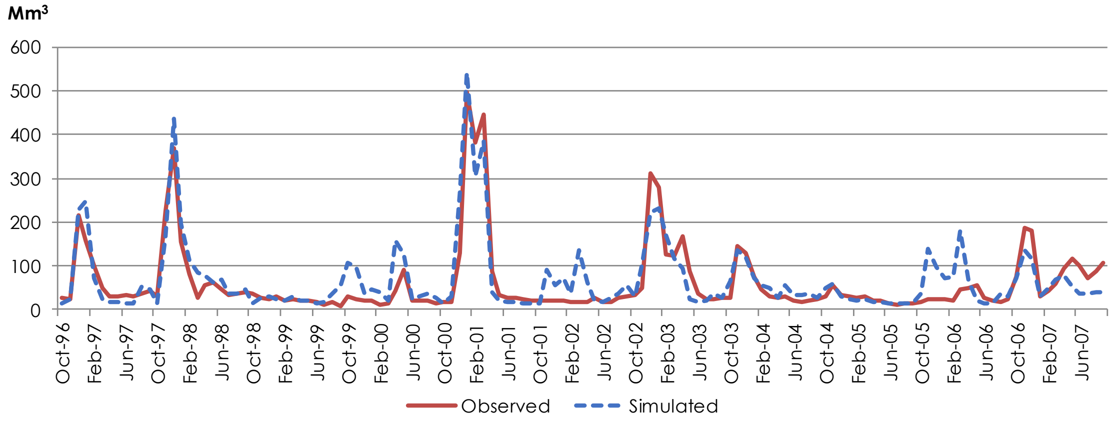

Figure 8. Comparison of observed and simulated flows upstream La Almendra reservoir.

Scenarios

By applying the proposed modelling framework to the TRB we want to illustrate the type of results produced in a clear-cut way, and to demonstrate that the final value of the service is sensitive to changes in the landscape and, more importantly, in water management. Hence, the assessment is performed under the business as usual scenario and two change scenarios: land use change and water management change. There are many possible changes that can be analysed under these broad scenarios, but we define extreme variations to obviously demonstrate the impact of both types of changes on the FPS.

- Business as usual: The baseline situation for land use and water management is considered.
- Land use change: It consists in the urbanisation of the Tormes headwaters sub-watershed which is originally mostly covered by natural vegetation. It is represented in the RRM through the reduction of evapotranspiration and infiltration (Yang et al., 2012). A constant reduction was applied along the simulated period, being 40% for the evapotranspiration and 10% for the infiltration.
- Water management change: This scenario proposes introducing a drastic change in the water management of the TRB by means of voiding Santa Teresa reservoir. This can be easily done in the WAM SIMGES by setting to 0 the storage capacity of the reservoir.

Results and discussion

Scenario 1: Business as usual

Considering the baseline conditions for land use and water management in the TRB, the Tormes headwaters sub-watershed produces the largest water volume that represents 72.7% of the total water resource generation on average, followed by the Snow melting sub-watershed with 24.4% of water production, the Middle tributaries that supply 1.6% of total runoff, and the Lower tributaries which produce 1.2%. These results, together with the configuration of the system lead to the distribution of water supply from each sub-watershed calculated by the water tracer and presented in Figure 9. It can be observed that water supply to all uses remains constant, matching the annual demand for water every simulated year except for the hydrological years 1980 and 1981 when all uses suffer from some supply deficit. Given the higher supply priority of Salamanca City, it has the lowest deficit which only represents 3% of its annual water demand in 1980. The irrigation uses have supply deficits around 18% and 2% of their corresponding annual demands in 1980 and 1981, respectively.

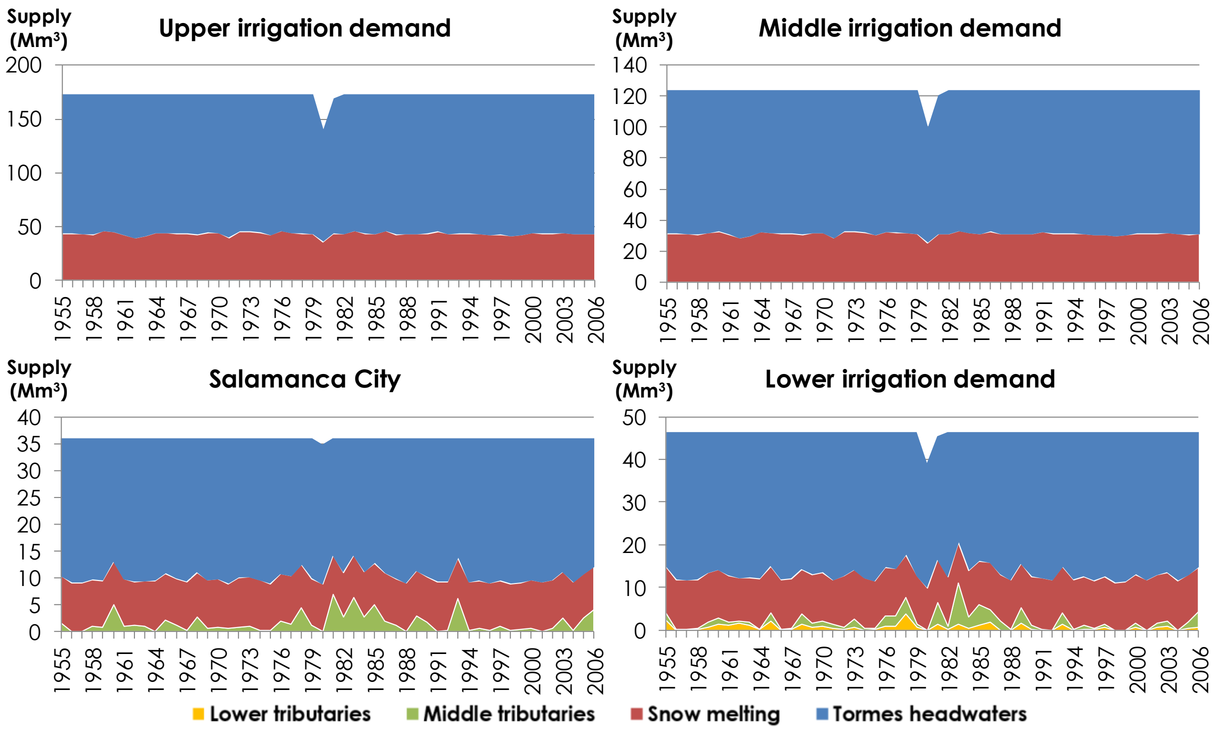

Figure 9. Water supply to the TRB water uses from each sub-watershed for scenario 1.

The annual value of the FPS in the TRB reaches 175.2M€ throughout the analysed period, except for the years with deficit in which the value falls to 171M€ in 1980 and 174.9M€ in 1981 (Figure 10). The proportion of value provided by each sub-watershed (72.6%, 24.6%, 2.7%, and 0.02% for the Tormes headwaters, Snow melting, Middle tributaries and Lower tributaries sub-watersheds respectively) is very similar to the fraction of water resources they produce. However, the utilisation of the water tracer allows identifying that the relative importance of the Middle tributaries increases in the economic valuation since they provide a significant amount of water to the urban use that assigns a higher value to water resources than agricultural uses.

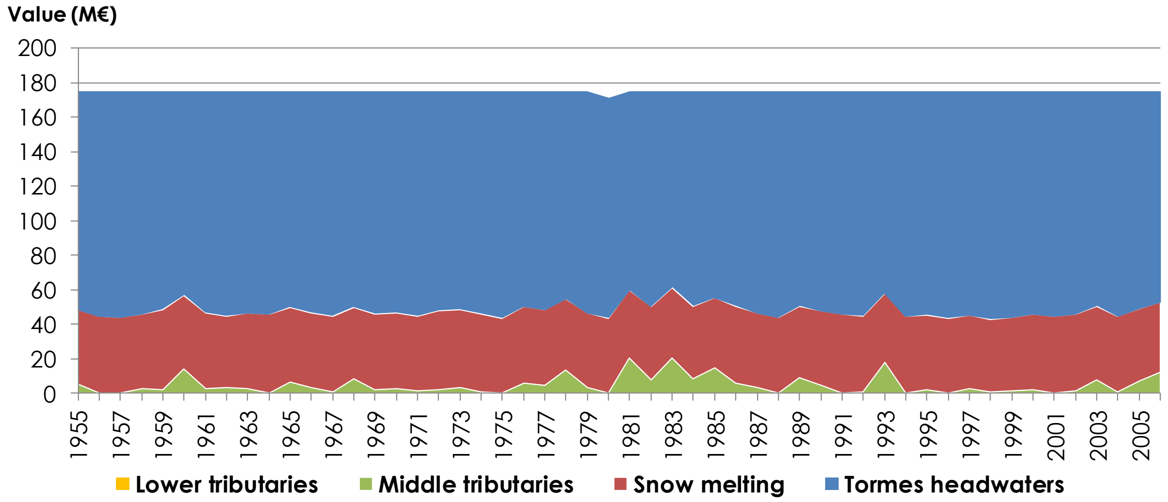

Figure 10. Annual series of the FPS economic value and contribution of each sub-watershed in scenario 1.

Scenario 2: Land use change of the Tormes headwaters sub-watershed

The urbanisation of the Tormes headwater sub-watershed makes the water resources produced by the Tormes headwaters rise from 427.8Mm^3 to 463.0Mm^3, whilst the water generated in the other sub-watersheds remains constant. The observed increase in water production due to land use transformation from natural vegetation to urban is in line with other studies (Bao and Fang, 2007; Du et al., 2012; Wagner et al., 2013).

As shown in Figure 11, the effect of the land use change on the water supply is that supply deficits in 1980 and 1981 are null or nearly zero. This is due to the fact that the water resources of the Tormes headwaters are generated upstream all water demands and, thus, they benefit from more water available. If the annual water supply varies, the economic value of the FPS also changes (Figure 12). In this scenario, the value of the service in 1980 and 1981 increases with respect to the baseline situation, being the augmentation of 4.2M€ and 0.3M€ in 1980 and 1981, respectively. The distribution pattern of water resources along the river system is also affected by the increase in the Tormes headwaters production, and so is the fraction of water that reaches each water use from each sub-watershed. This results in a different distribution of value among the sub-watersheds. In this case, the Tormes headwaters sub-watershed is responsible for 74.5% or FPS value, the Snow melting sub-watershed provides 23.2% of the value, 2.3% corresponds to the Middle tributaries, and 0.02% to the Lower tributaries.

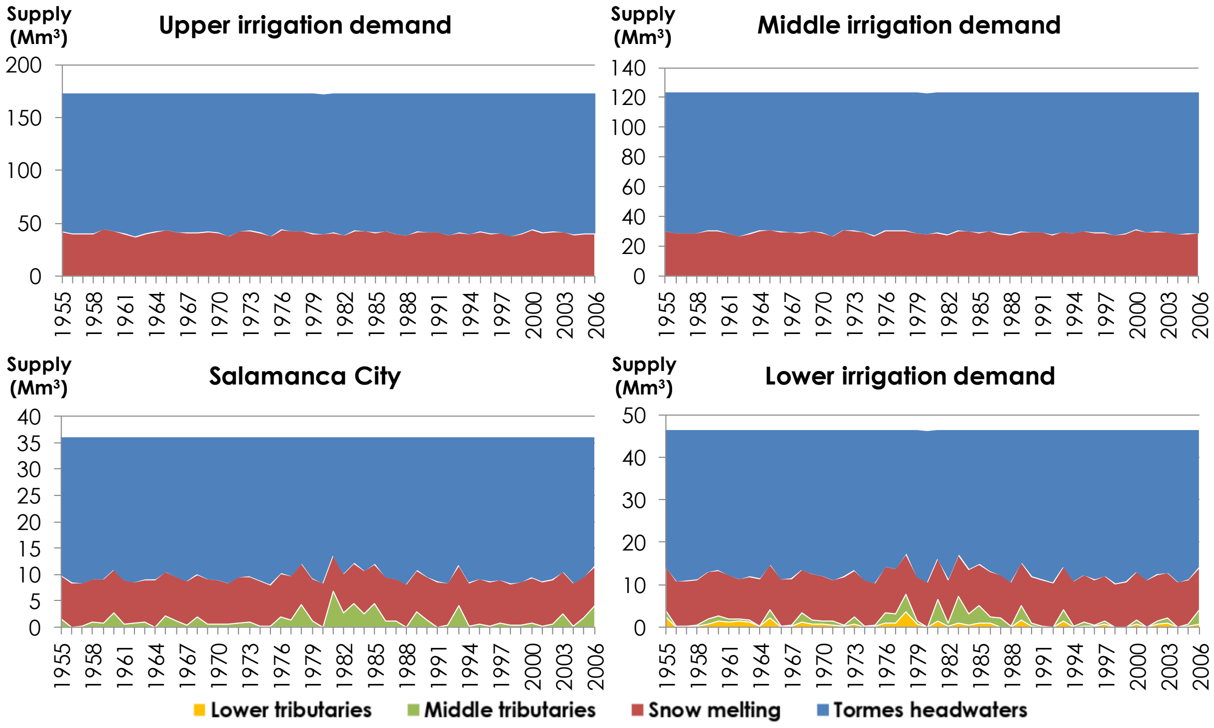

Figure 11. Water supply to the TRB water uses from each sub-watershed for scenario 2.

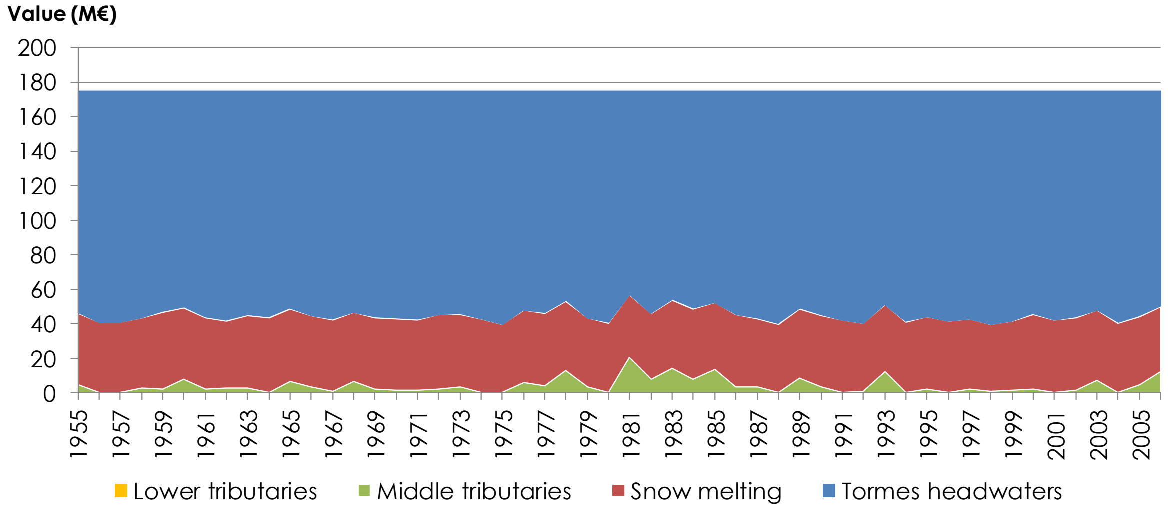

Figure 12. Annual series of the FPS economic value and contribution of each sub-watershed in scenario 2.

Scenario 3: Water management change

This modification of water management or infrastructures does not affect the runoff generation by the different sub-watersheds with respect to scenario 1. Nonetheless, as depicted in Figure 13, the impact on the water supply is huge due to the lack of regulation capacity of the water resources provided by the most productive sub-watersheds (i.e. Tormes headwaters and Snow melting). In this scenario, the only water use with an acceptable level of water supply with respect to its demand is Salamanca City because it has a high supply priority. On the contrary, the irrigation uses barely get to 40% of their annual demand most of the time.

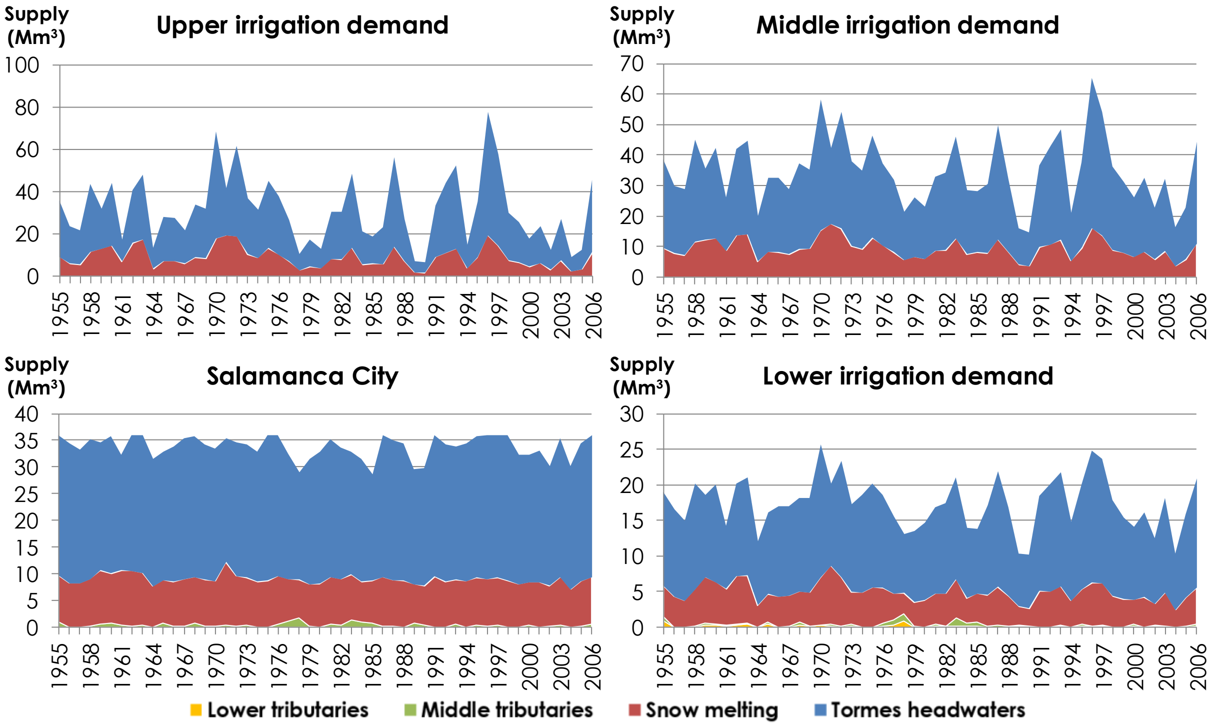

Figure 13. Water supply to the TRB demands from each sub-watershed for scenario 3.

When the supply values are translated into economic benefits by means of the demand curves, the result is an average annual reduction in the FPS benefit of 29.7M€. The relative contribution of the sub-watersheds to the total value of the service remains almost unchanged with respect to scenario 1. Nevertheless, the Tormes headwaters and the Snow melting sub-watersheds slightly increase their benefit provision (72.8% and 26.3%, respectively) by partly replacing the Middle tributaries (0.9%) in the supply to Salamanca City. This is because Salamanca City proportionally receives more water resources from the Tormes headwaters and the Snow melting sub-watersheds, as they cannot be stored to be used in low flow periods. It is interesting to notice that the year with the lowest economic value of the service in this scenario (1990) does not coincide with the baseline scenario (see Figure 10 and Figure 14) in which the lowest benefit was coincident with the driest year (1980). The explanation can be found in the monthly results presented in Figure 15. Even though the accumulated runoff from October 1980 to September 1982 is lower than the runoff from October 1989 to September 1991, the flows during the dry season are lower in the later period, and cause higher supply deficits to the irrigation demands. This effect is buffered by the existence of the reservoir in scenario 1 but not in scenario 3.

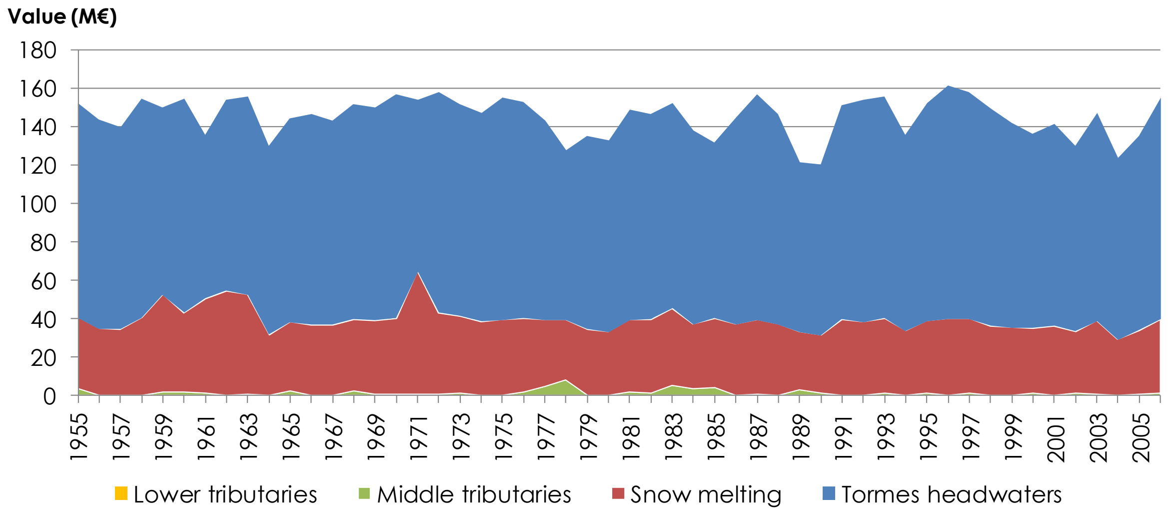

Figure 14. Annual series of the FPS economic value and contribution of each sub-watershed in scenario 3.

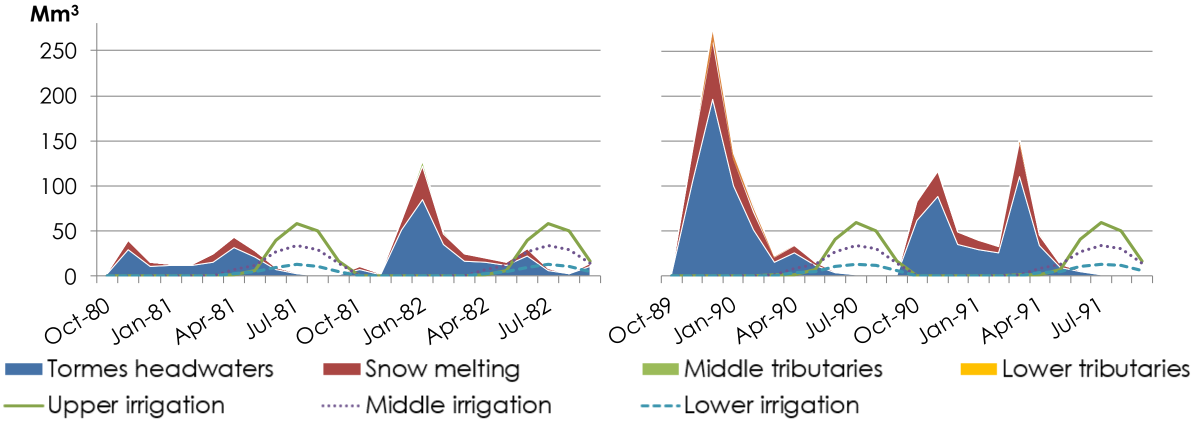

Figure 15. Monthly comparison of the water resources produced and the water demands of the irrigation uses.

General discussion

Tormes headwaters is the most productive sub-watershed from the water quantity and the economic perspectives, followed by the Snow melting sub-watershed. The Middle tributaries are relevant to ensure a high supply reliability to the urban use in scenarios 1 and 2; especially during the drought episode in which it provides most of the required water for some months while the upstream resources are stored in the reservoir (Figure 16). Finally, the Lower tributaries play a minor role given that they are located at the end of the system and can only be used by the Lower irrigation demand.

Due to the configuration of the TRB infrastructures, each water demand can only use water from the upstream sub-watersheds. If there were conveyance infrastructures to carry water and make it available upstream, the numbers would vary.

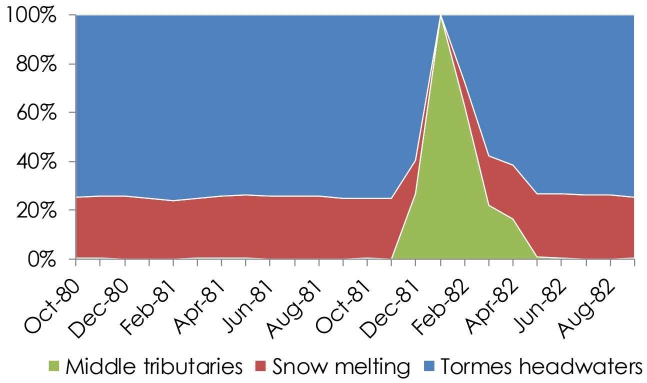

Figure 16. Monthly fraction contributed by each sub-watershed to Salamanca City in scenario 1.

The scenario analysis demonstrates the high influence that water management has on the FPS. The level of detail and accuracy that the WAM provides regarding water infrastructures (e.g reservoirs and transport networks) and management rules (e.g. supply priorities and inter-annual regulation) cannot be obtained with the existing ES tools. The last scenario is probably the most interesting since it clearly shows the influence of water management and temporal variability on the delivery of the service, which is precisely the advantage of using IWRM models for freshwater ES assessment instead of ES tools as pointed in the introduction.

The comparison across scenarios and along time in each scenario, reveals that the value of the service falls when the water supply decreases. This fact can be confusing, given that the economic theory states that when a resource becomes scarce, its value increases. As reflected by the demand curves in Figure 6, the unitary value of water indeed increases when the supply diminishes. This increase is not constant and, depending on the magnitude of the supply deficit, the total economic value of the water supply may decrease.

The monthly time scale appears to be appropriate to capture seasonal variability of water resources (see Figure 8), water demands and their interaction (see Figure 15). In fact, some of the analysed aspects in the application to the TRB would have been disguised had the time step been larger. A clear example is the occurrence of the lowest economic value of the FPS in scenario 3. Had the simulations been performed at annual scale, it would have occurred in 1980 since the annual gap between water availability and demand is the largest. However, the monthly mismatch between water availability and demand is higher in 1990. Finally, the water tracer ensures that the mapping of the results reflects the real contribution of each watershed to the value of the FPS, including cases in which there are returns from demands. Although not applied in the case study for the sake of simplicity, the possibility to represent the effect of inter-basin water transfers that modify the natural movement of water along the river system or groundwater recharge, regulation and exploitation is a valuable aspect of the proposed modelling framework.

Some difficulties or limitations for the application of this methodology come from data acquisition. Demand functions are the most rigorous way to conduct a marginal economic valuation. However, they are not commonly produced due to the cost of the required studies; and, if generated, they are aggregated at regional scale, instead of detailed for each water use. It is important to notice that valuation techniques face limitations that are as yet unresolved. Consequently, decision makers should interpret and use valuation data with caution (The Economics of Ecosystems & Biodiversity, 2010). Another drawback is the lack of information about the modification of the parameters of the models (mainly the RRM) to represent changes introduced in each scenario, such as land use changes, which forces the adoption of simplifications and assumptions that go against the quality of the final output. However, problems with data are not specific for the modelling framework proposed here; in fact, they are common to all models.

Finally, it is important to point out the relevance of applying the ES approach in a broad sense by considering all the potential ES affected (or most of them) as this is a relevant source of uncertainty (Boithias et al., 2016). A good example for this is the result obtained in scenario 2, in which land use changes from natural vegetation to urban cover led to the improvement of the FPS. Reasonably, this type of land use change would negatively affect many other ES, and a global ES assessment would probably indicate that this action worsens the state of ecosystems and their productivity. Similarly, the removal of the Santa Teresa reservoir in scenario 3 implies the loss of FPS, but other ES value would increase due to the gains in longitudinal connectivity in the river. In this regard, the methodology presented here aims to contribute to part of the overall ES analysis.

Conclusions

This paper proposes a modelling framework which links three models, commonly used in IWRM, and economic data to quantify, value and map the FPS with detailed consideration of water management rules and infrastructures. Results from the application to the TRB show that the FPS is sensitive to land and water management changes. Actions affecting the landscape have an effect on the ecosystems which provide the service and, consequently, they modify the amount of water produced by each water source. This brings the variation of the economic value of the FPS, even if water management practices are identical. On the other hand, measures that modify the water management do not have any influence on the landscape ecosystems and, thus, do not affect the water yield of water sources. Nevertheless, these kinds of measures modify the economic value of the service by changing the distribution pattern of water resources along the river system and the water supply to the different uses. Hence, it is extremely important to faithfully represent water management practices when assessing the FPS. Furthermore, bouncing off the value of the service from water uses to water sources provides helpful information in order to protect the main sources of water in a river basin.

As a general conclusion, we can say that IWRM models are able to represent the main processes involved in the provision of FPS reflecting the effects of management actions and providing temporally and spatially detailed results. Decision support systems for IWRM offer sets of interconnected models which can be sequentially run to derive results in terms of water-related ES with slight adaptation. This contributes to advance towards the real implementation of the ecosystem approach by helping to understand the multiple effects of management and policy changes on ecosystems.

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Improved modelling of the freshwater provisioning ecosystem service in water scarce river basins

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