

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

Andrea Momblanch^{a,b}, Javier Paredes-Arquiola^a, Joaquín Andreu^a

^a Research Institute of Water and Environmental Engineering, Universitat Politècnica de València,
Spain

^b Cranfield Water Science Institute, Cranfield University, United Kingdom

Corresponding author email: andrea.momblanch-benavent@cranfield.ac.uk

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

29 **Introduction**

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

46 In the case of water resources, the management scale is the river basin as established by the
47 European Water Framework Directive (European Parliament and Council, 2000) and in line with the
48 Integrated Water Resources Management paradigm (Global Water Partnership, 2000). Even though
49 water is essential for most ecosystem processes that rely on water abundance, temporal and spatial
50 distribution, there are only two types of ES that are related to its management. Aquatic ES account
51 for the benefits provided by freshwater ecosystems such as water purification (Keeler et al., 2012; La
52 Notte et al., 2012; Liqueste et al., 2011; Terrado et al., 2016) and habitat for fish (Liqueste et al., 2016;
53 Sample et al., 2016). On the other hand, hydrologic ES describe the benefits to people derived from
54 the relationship between terrestrial ecosystems and freshwater quantity and quality (Brauman,
55 2015); some examples are freshwater provision (Boithias et al., 2014; Dennedy-Frank et al., 2016;
56 Guo et al., 2000; Karabulut et al., 2016; Terrado et al., 2014), flood mitigation (Fu et al., 2013; Watson
57 et al., 2016) and pollution abatement (Bogdan et al., 2016; Fu et al., 2012).

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

73 Bearing the above in mind, the selection of the model to assess hydrologic ES in water scarce river
74 basins should be thorough. Simulation models especially designed for ES assessment, or ES tools,
75 integrate ecological and economic aspects for several ES considering their spatial variability (Bagstad
76 et al., 2013a). They allow analysing tradeoffs between ES under different scenarios and are attainable
77 for non-experts (Terrado et al., 2014). An extensive review of ES tools can be found in (Bagstad et al.,
78 2013a). The Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST) (Tallis et al., 2013) is
79 likely the most widely known ES tool. It is a spatially explicit model to estimate levels of different ES
80 benefits in a static timeframe, usually an average year (Terrado et al., 2016). InVEST includes
81 freshwater provisioning, sediment retention, and water purification as hydrologic ES. It accounts for
82 the processes taking place in the landscape considering simplified hydrological relationships whose
83 main input are land use-land cover maps linked to biophysical parameters such as roots depth and
84 retention capacity of vegetation. The instream processes are also simplified and limited to the
85 conveyance of water to its use location, without regarding the influence of water infrastructures and
86 their operation.

87 Another well-known ES tool is the web-based Artificial Intelligence for Ecosystem Services (ARIES)
88 (Villa et al., 2014). It applies a probabilistic Bayesian network approach which uses a library of models

89 and spatial data to quantify ES flows and uncertainty when little data is available (Bagstad et al.,
90 2013b), but it also allows employing biophysical relationships when enough data is accessible
91 (Vigerstol and Aukema, 2011). The hydrologic ES addressed by ARIES are flood regulation, nutrient
92 regulation, sediment regulation, and water supply. It works with a time step ranging from hours to
93 years, and does not value the ES in economic units (Villa et al., 2014). Even though this ES tool is
94 flexible to introduce instream processes, it lacks the capabilities to faithfully represent water
95 management influence on the delivery of hydrologic ES. Moreover, the model complexity can hinder
96 the understanding of the modelled processes and the results for decision makers and stakeholders
97 (Vigerstol and Aukema, 2011).

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

114 This paper focuses on the assessment of the Freshwater Provisioning hydrologic ES (FPS). Brauman
115 et al. (2007) define it as the natural process that modifies the quantity of water for extractive (e.g.
116 drinking, irrigation and industrial uses) and on site purposes (e.g hydropower generation, water
117 recreation and transport). The main aim of the study is proposing a modelling framework composed
118 of IWRM models to assess the FPS with detailed consideration of water resources management
119 impacts. The paper describes the linkage and adaptation of a rainfall-runoff model, a water allocation

120 model and a water quality model to obtain the spatial distribution of the FPS in biophysical and
121 economic units. To the best knowledge of the authors, a similar modelling approach has not been
122 presented previously. The methodology is illustrated in the Tormes River Basin (TRB) in Spain, which
123 has a predominant semi-arid climate, for two scenarios that introduce changes in the landscape and
124 in water management with respect to the business as usual. Results demonstrate the influence of
125 water management on the delivery of the service, which justifies the convenience of using IWRM
126 models to make up for the limitations of ES tools in water scarce river basins.

127 **Material and Methods**

128 Modelling framework

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

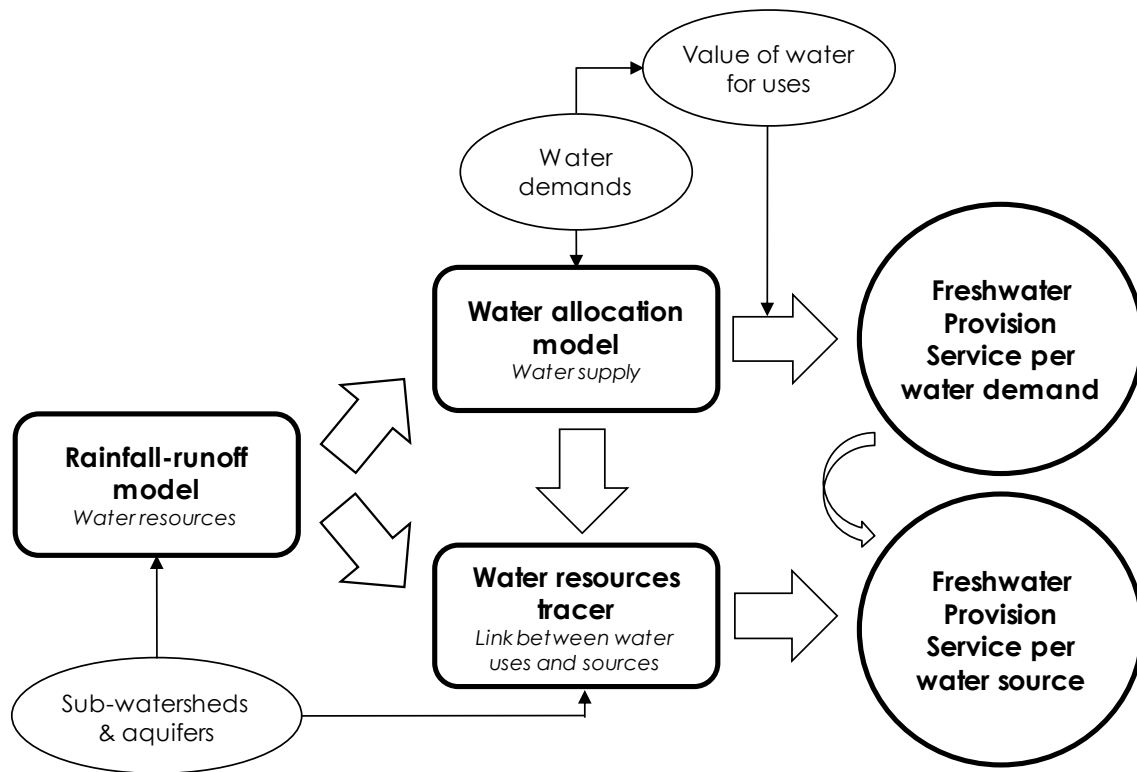
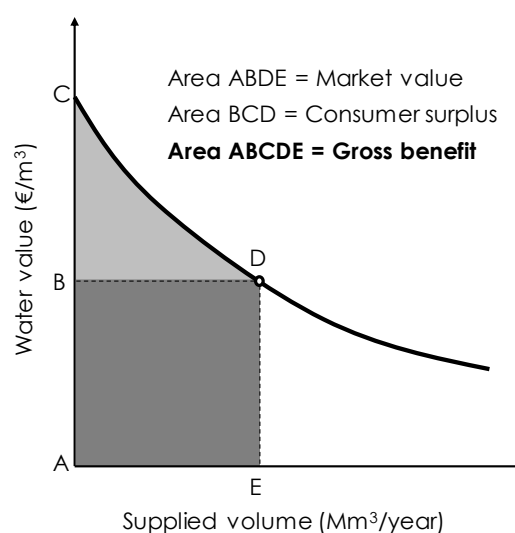


Figure 1. Modelling framework for the assessment of the FPS.

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

159 able to represent common water management strategies such as water supply priorities and
160 operation rules.

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

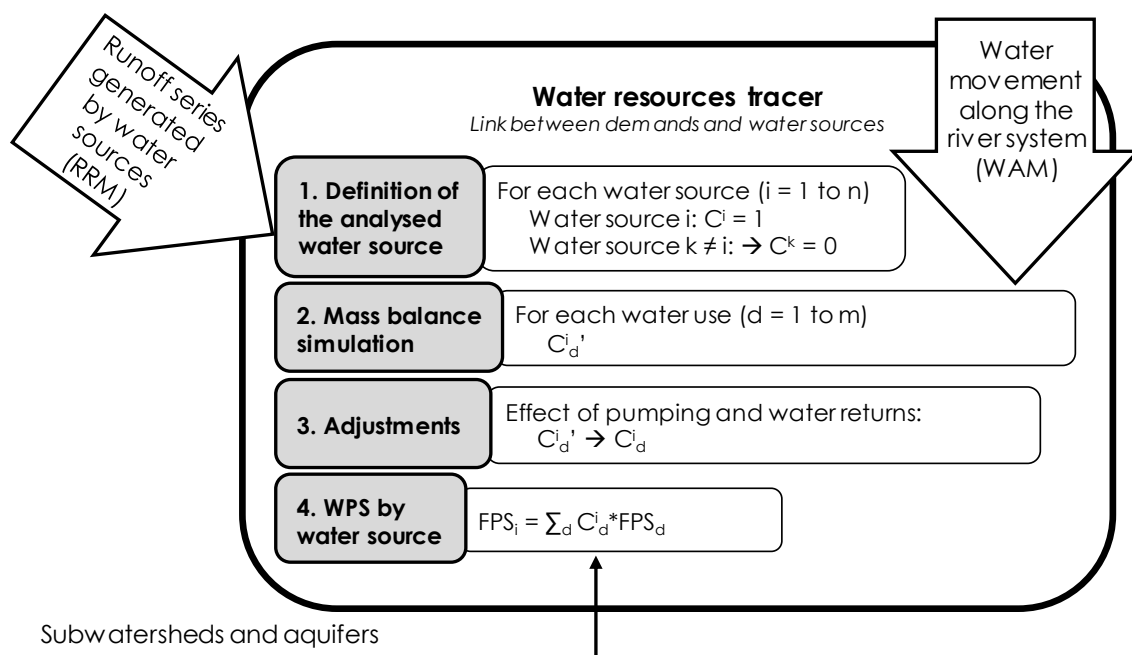


175
176 **Figure 2. Obtaining of the gross benefit from a water demand curve.**

177 Since the water supplies provided by the WAM have a monthly step, they are yearly accumulated to
178 be compatible with the demand curves. The annual benefit resulting from the demand curves for
179 each water use is then temporally distributed according to the monthly water supply. The total

180 monthly benefit provided by the FPS in the whole river basin is calculated as the sum of the monthly
 181 benefits of all water uses. These results are helpful when analysing different water management
 182 strategies.

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



198

199

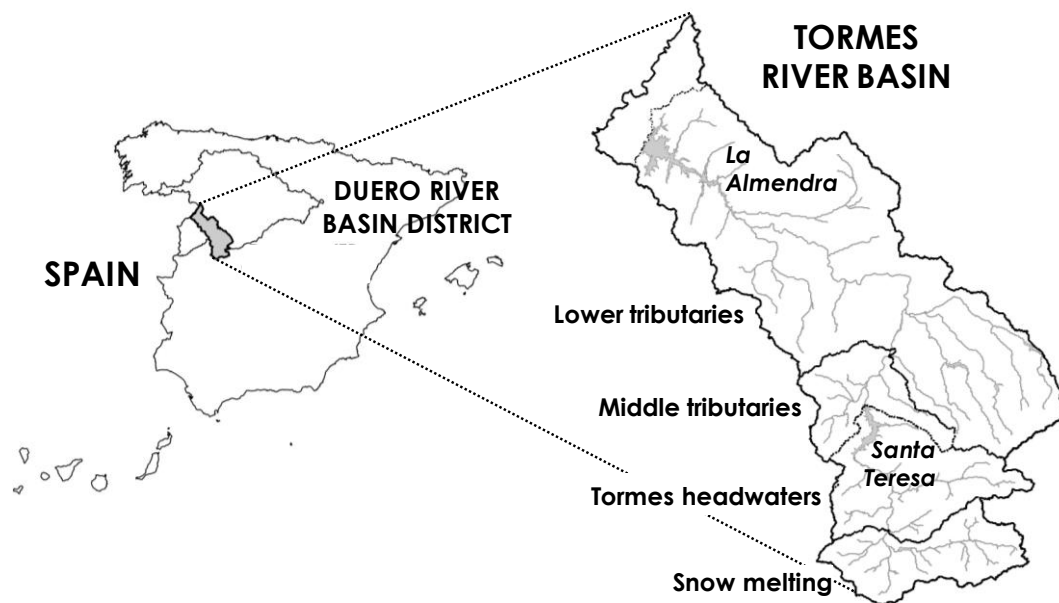
Figure 3. Water tracer diagram for the mapping of the FPS.

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

222 Study area: Tormes River Basin

223 The TRB belongs to the Duero River Basin District in Spain (see Figure 4). It covers an area of 9,568km²
 224 with an average precipitation of 529.9mm/year and a potential evapotranspiration of
 225 826.28mm/year, resulting in a mean annual total runoff of 1,678.2Mm³. It has a predominant semi-
 226 arid climate with Mediterranean and Continental influence. The TRB spans from the mountainous
 227 region of Sierra de Gredos and flows north-west until the convergence with the Duero River, just
 228 downstream La Almendra reservoir. It counts with large Natura 2000 sites at the heading and at the
 229 lower part of the basin.

230 The main water uses in the TRB are agriculture with a water demand of 319.5Mm³/year, urban
231 demands that amount to 38.9Mm³/year, and hydropower uses that are mostly run-of-river stations
232 and, hence, do not determine water management. The total population in the TRB is around 280,000
233 inhabitants of which more than 160,000 live in the city of Salamanca. Even though the basin holds
234 several reservoirs, only Santa Teresa performs inter-annual regulation since La Almendra reservoir
235 only serves downstream uses, which are outside the TRB. The conceptualisation of the basin
236 considered in this application is a simplification of the real system. This is because the purpose of the
237 application case is not getting insight of the real behaviour of the TRB, but exemplifying the type of
238 analysis that the modelling framework allows in a simple and clear way. The simplified TRB only
239 contains the urban demand of Salamanca with the highest supply priority, the irrigation uses grouped
240 in three areas with equal supply priority, Santa Teresa reservoir, and the inflows generated by all sub-
241 watersheds grouped into four (see Figure 4 and Figure 5).



242
243 **Figure 4. Location of the TRB, main reservoirs and sub-watersheds.**

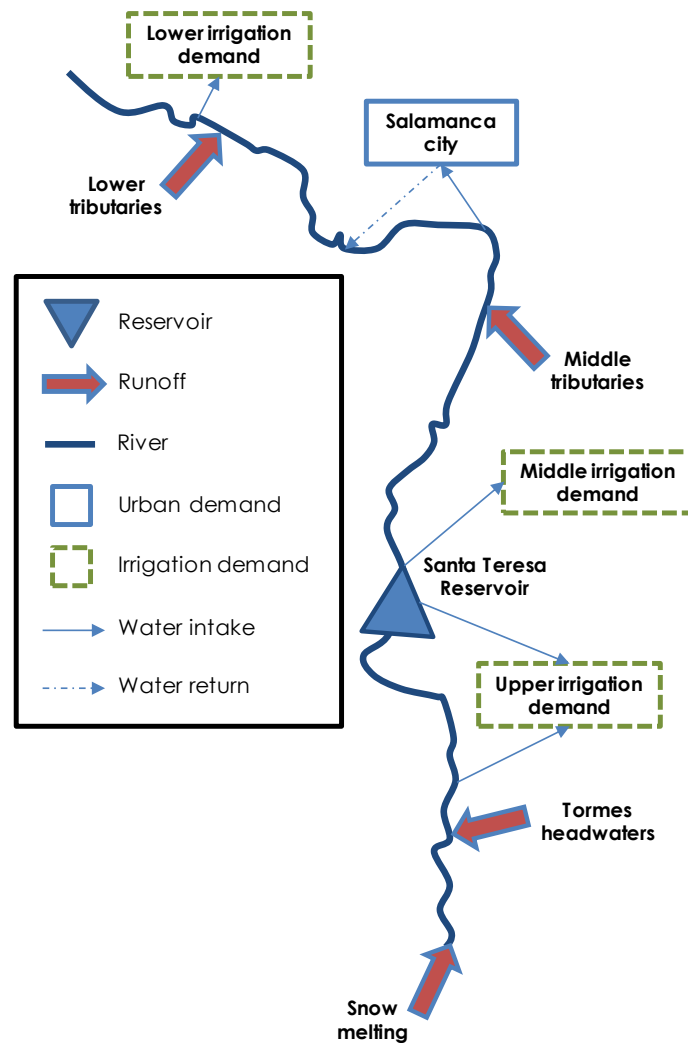
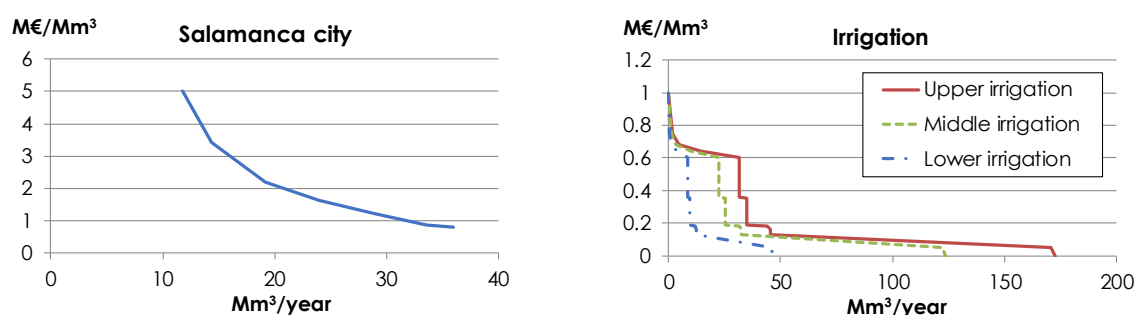


Figure 5. Diagram of the simplified TRB.

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

258 from the former, while the demand curves for the three irrigation areas are obtained from the latter
 259 (Figure 6).



260

261

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

262 IWRM tools

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

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

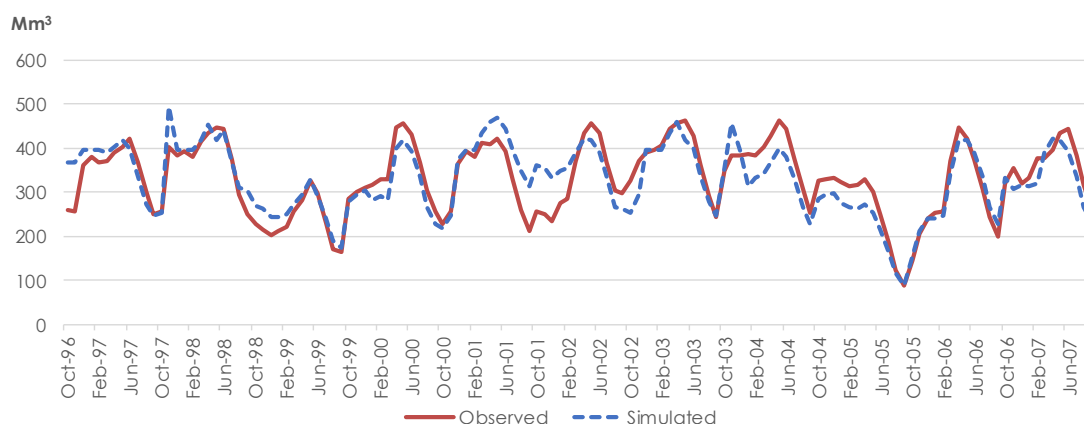


Figure 7. Comparison of observed and simulated volumes stored in Santa Teresa reservoir.

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.

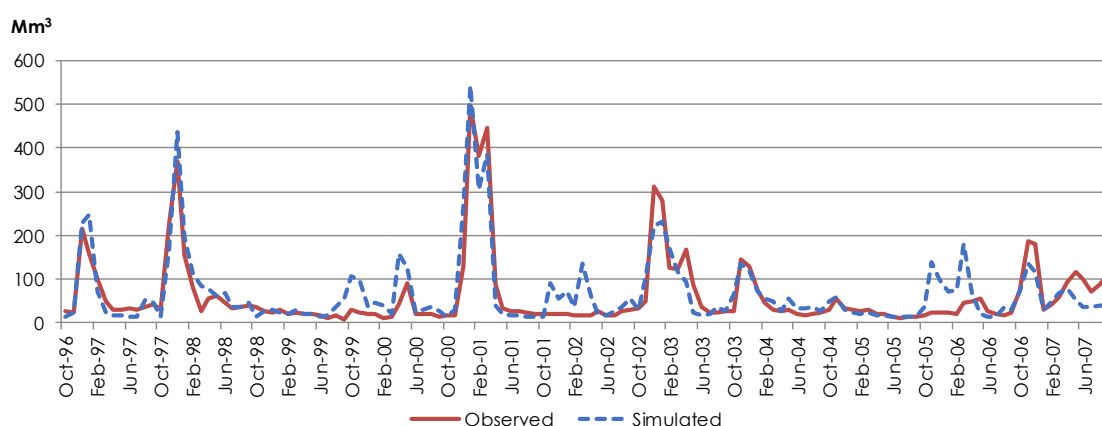


Figure 8. Comparison of observed and simulated flows upstream La Almendra reservoir.

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.

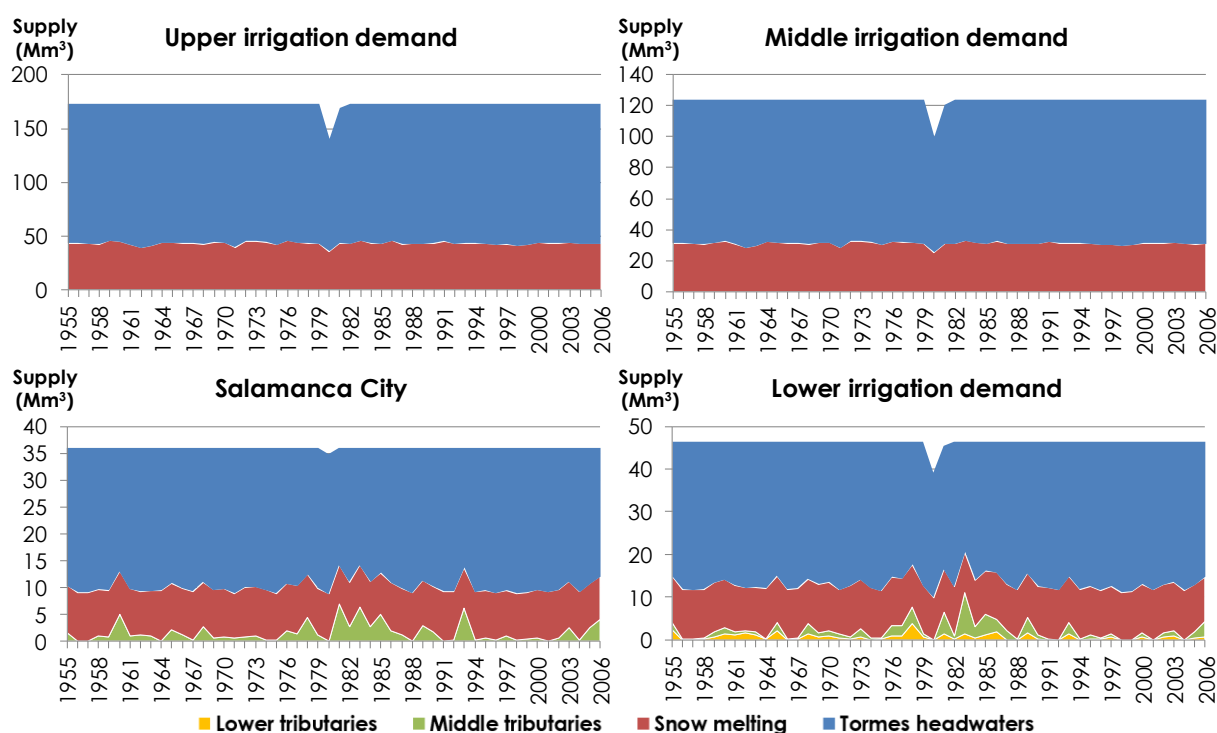


Figure 9. Water supply to the TRB water uses from each sub-watershed for scenario 1.

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.

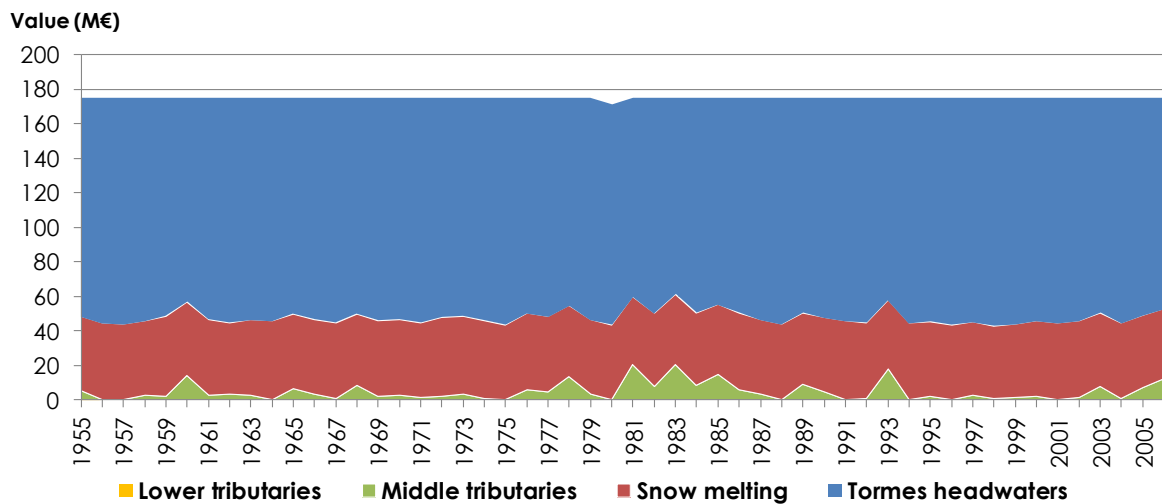


Figure 10. Annual series of the FPS economic value and contribution of each sub-watershed in scenario 1.

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 427.8Mm³ to 463.0Mm³, 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% of 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.

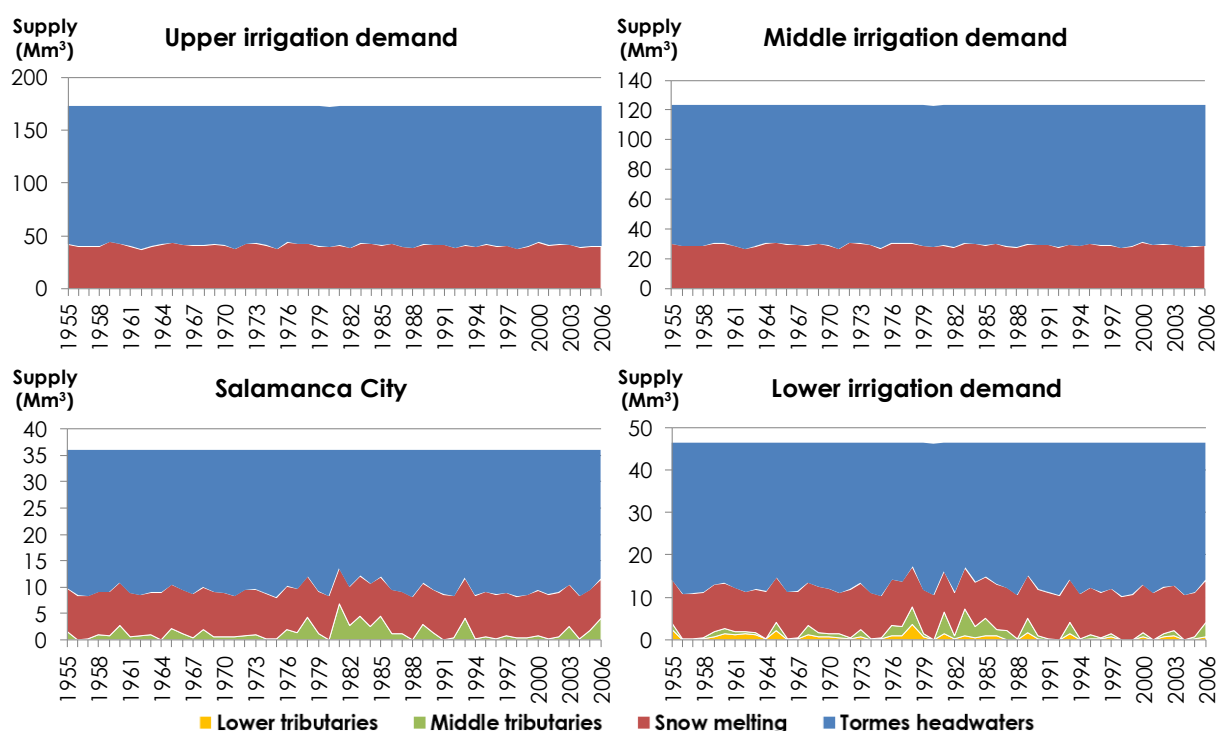


Figure 11. Water supply to the TRB water uses from each sub-watershed for scenario 2.

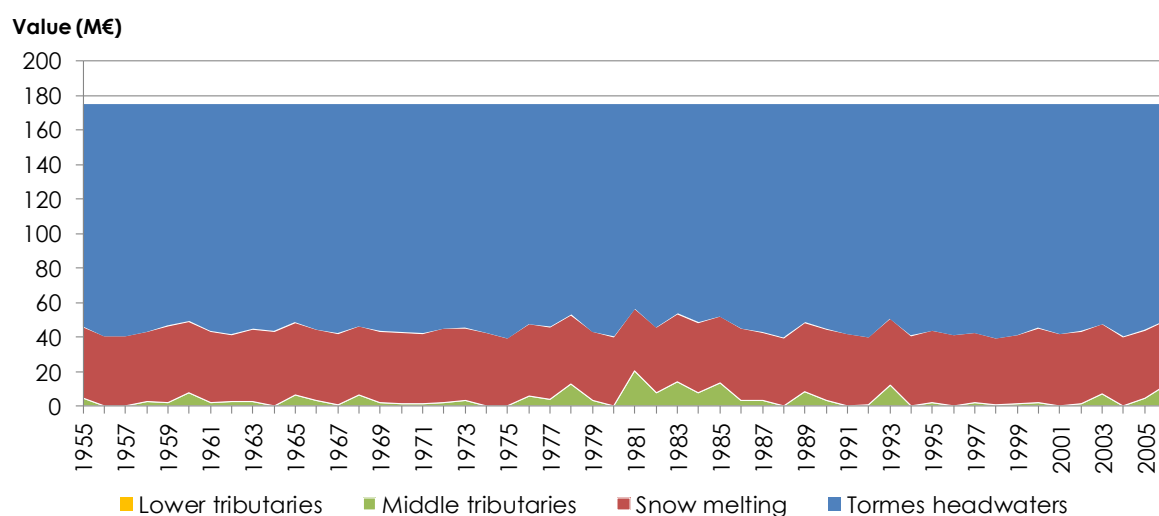
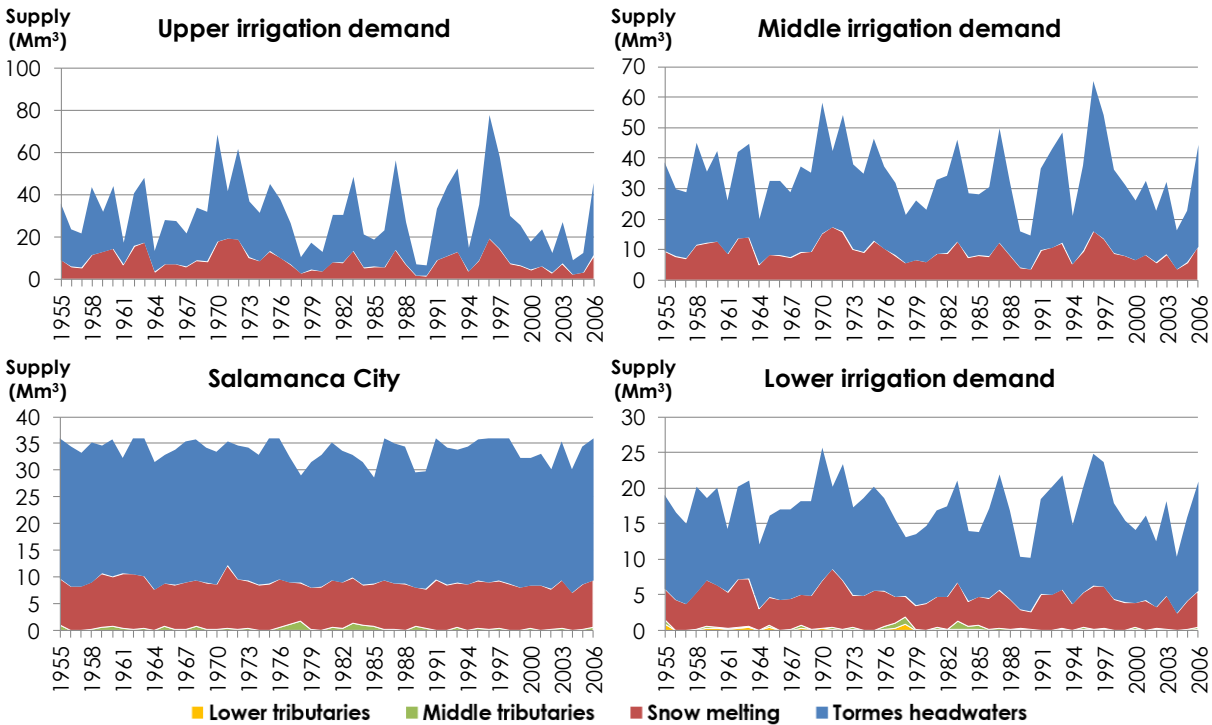


Figure 12. Annual series of the FPS economic value and contribution of each sub-watershed in scenario 2.

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

360 scenario, the only water use with an acceptable level of water supply with respect to its demand is
 361 Salamanca City because it has a high supply priority. On the contrary, the irrigation uses barely get
 362 to 40% of their annual demand most of the time.



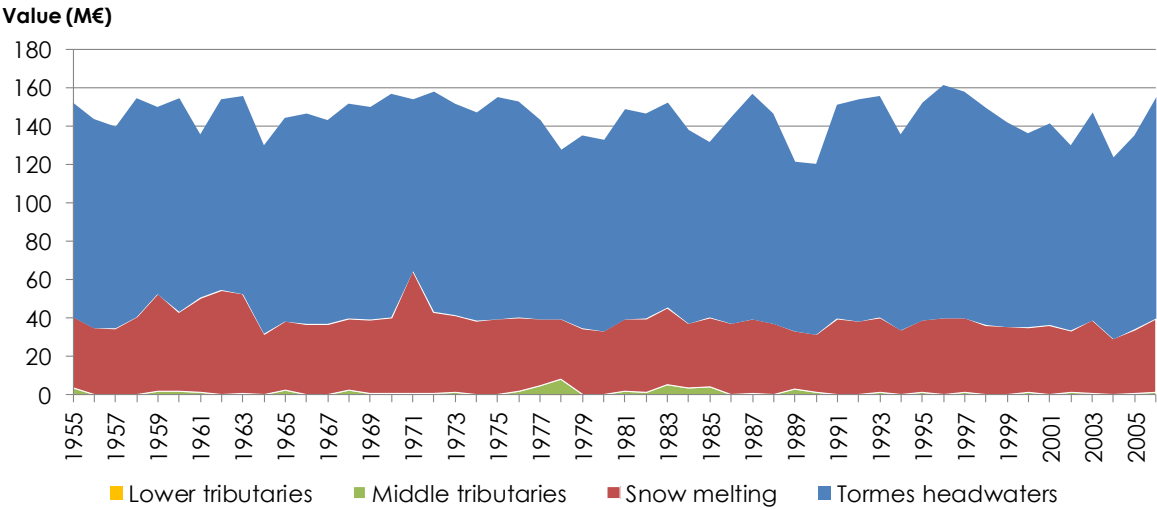
363

364

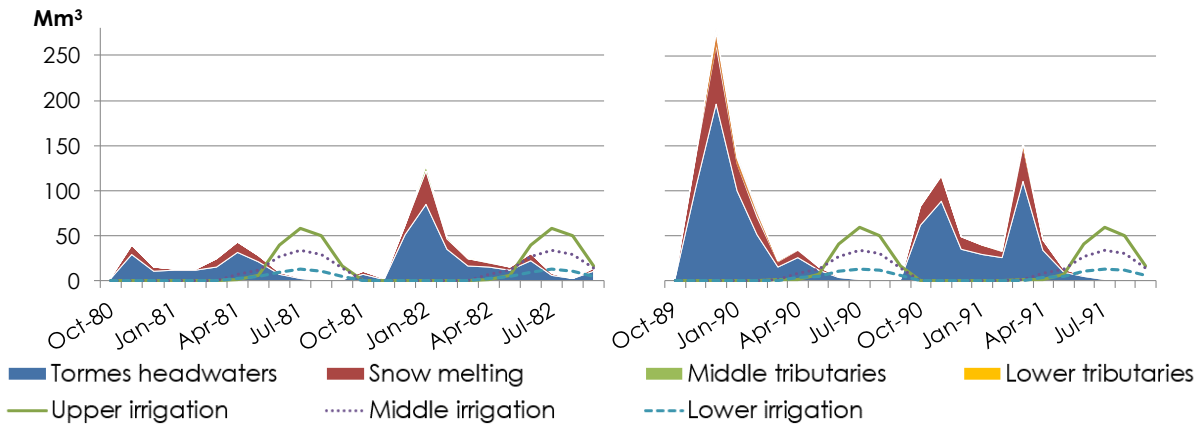
Figure 13. Water supply to the TRB demands from each sub-watershed for scenario 3.

365 When the supply values are translated into economic benefits by means of the demand curves, the
 366 result is an average annual reduction in the FPS benefit of 29.7M€. The relative contribution of the
 367 sub-watersheds to the total value of the service remains almost unchanged with respect to scenario
 368 1. Nevertheless, the Tormes headwaters and the Snow melting sub-watersheds slightly increase their
 369 benefit provision (72.8% and 26.3%, respectively) by partly replacing the Middle tributaries (0.9%) in
 370 the supply to Salamanca City. This is because Salamanca City proportionally receives more water
 371 resources from the Tormes headwaters and the Snow melting sub-watersheds, as they cannot be
 372 stored to be used in low flow periods. It is interesting to notice that the year with the lowest economic
 373 value of the service in this scenario (1990) does not coincide with the baseline scenario (see Figure 10
 374 and Figure 14) in which the lowest benefit was coincident with the driest year (1980). The explanation
 375 can be found in the monthly results presented in Figure 15. Even though the accumulated runoff
 376 from October 1980 to September 1982 is lower than the runoff from October 1989 to September
 377 1991, the flows during the dry season are lower in the later period, and cause higher supply deficits

378 to the irrigation demands. This effect is buffered by the existence of the reservoir in scenario 1 but
 379 not in scenario 3.



380
 381 **Figure 14. Annual series of the FPS economic value and contribution of each sub-watershed in scenario 3.**



382
 383 **Figure 15. Monthly comparison of the water resources produced and the water demands of the irrigation**
 384 **uses.**

385 General discussion

386 Tormes headwaters is the most productive sub-watershed from the water quantity and the economic
 387 perspectives, followed by the Snow melting sub-watershed. The Middle tributaries are relevant to
 388 ensure a high supply reliability to the urban use in scenarios 1 and 2; especially during the drought
 389 episode in which it provides most of the required water for some months while the upstream
 390 resources are stored in the reservoir (Figure 16). Finally, the Lower tributaries play a minor role given
 391 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.

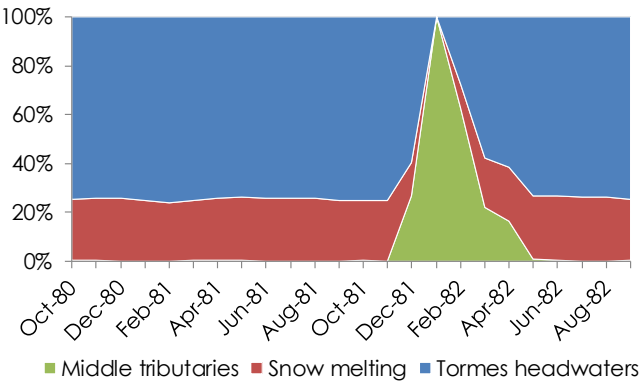


Figure 16. Monthly fraction contributed by each sub-watershed to Salamanca City in scenario 1.

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

415 between water availability and demand is the largest. However, the monthly mismatch between
416 water availability and demand is higher in 1990. Finally, the water tracer ensures that the mapping
417 of the results reflects the real contribution of each watershed to the value of the FPS, including cases
418 in which there are returns from demands. Although not applied in the case study for the sake of
419 simplicity, the possibility to represent the effect of inter-basin water transfers that modify the natural
420 movement of water along the river system or groundwater recharge, regulation and exploitation is a
421 valuable aspect of the proposed modelling framework.

422 Some difficulties or limitations for the application of this methodology come from data acquisition.
423 Demand functions are the most rigorous way to conduct a marginal economic valuation. However,
424 they are not commonly produced due to the cost of the required studies; and, if generated, they are
425 aggregated at regional scale, instead of detailed for each water use. It is important to notice that
426 valuation techniques face limitations that are as yet unresolved. Consequently, decision makers
427 should interpret and use valuation data with caution (The Economics of Ecosystems & Biodiversity,
428 2010). Another drawback is the lack of information about the modification of the parameters of the
429 models (mainly the RRM) to represent changes introduced in each scenario, such as land use changes,
430 which forces the adoption of simplifications and assumptions that go against the quality of the final
431 output. However, problems with data are not specific for the modelling framework proposed here;
432 in fact, they are common to all models.

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

442 **Conclusions**

443 This paper proposes a modelling framework which links three models, commonly used in IWRM, and
444 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.

Acknowledgments

The authors acknowledge the support of Universitat Politècnica de València through its Support Programme for Research and Development. We also wish to thank Confederación Hidrográfica del Duero (belonging to the Spanish Ministry of Agriculture, Food and Environment) for the data provided in developing this study and the Spanish Ministry of Economy and Competitiveness for its financial support through the projects SCARCE (Consolider-Ingenio 2010 CSD2009-00065) and NUTEGES (CGL2012-34978). We also value the support provided by the European Community in financing the Seventh Framework Program projects DROUGHT-R&SPI (FP7-ENV-2011, 282769), ENHANCE (FP7-ENV-2012, 308438), the H2020 project IMPREX (H2020-WATER-2014-2015, 641811), the grant WAMCD (EC-DG Environment No. 07.0329/2013/671291/ SUB/ENV.C.1) and the Life+ project LIFE ALBUFERA (LIFE12 ENV/ES/000685).

475 **References**

- 476 Andreu, J., Capilla, J., Sanchis, E., 1996. AQUATOOL, a generalized decision-support system for
477 water-resources planning and operational management. *J. Hydrol.* 177, 269–291.
478 doi:http://dx.doi.org/10.1016/0022-1694(95)02963-X
- 479 Arnold, J.G., Srinivasan, R., Muttiah, R.S., Williams, J.R., 1998. Large area hydrologic modeling and
480 assessment. Part I: Model development. *J. Am. Water Resour. Assoc.* 34, 73–89.
481 doi:http://dx.doi.org/10.1111/j.1752-1688.1998.tb05961.x
- 482 Bagstad, K.J., Semmens, D.J., Waage, S., Winthrop, R., 2013a. A comparative assessment of
483 decision-support tools for ecosystem services quantification and valuation. *Ecosyst. Serv.* 5,
484 27–39. doi:http://dx.doi.org/10.1016/j.ecoser.2013.07.004
- 485 Bagstad, K.J., Semmens, D.J., Winthrop, R., 2013b. Comparing approaches to spatially explicit
486 ecosystem service modeling: A case study from the San Pedro River, Arizona. *Ecosyst. Serv.* 5,
487 40–50. doi:http://dx.doi.org/10.1016/j.ecoser.2013.07.007
- 488 Bao, C., Fang, C., 2007. Water resources constraint force on urbanization in water deficient regions:
489 A case study of the Hexi Corridor, arid area of NW China. *Ecol. Econ.* 62, 508–517.
490 doi:http://dx.doi.org/10.1016/j.ecolecon.2006.07.013
- 491 Bergström, S., 1995. The HBV model, in: Singh, V.P. (Ed.), *Computer Models of Watershed*
492 *Hydrology*. Water Resources Publications, LLC, Highlands Ranch.
- 493 Bogdan, S.-M., Pătru-Stupariu, I., Zaharia, L., 2016. The Assessment of Regulatory Ecosystem
494 Services: The Case of the Sediment Retention Service in a Mountain Landscape in the Southern
495 Romanian Carpathians. *Procedia Environ. Sci.* 32, 12–27.
496 doi:http://dx.doi.org/10.1016/j.proenv.2016.03.008
- 497 Boithias, L., Acuña, V., Vergoñós, L., Ziv, G., Marcé, R., Sabater, S., 2014. Assessment of the water
498 supply: demand ratios in a Mediterranean basin under different global change scenarios and
499 mitigation alternatives. *Sci. Total Environ.* 470, 567–577. doi:10.1016/j.scitotenv.2013.10.003
- 500 Boithias, L., Terrado, M., Corominas, L., Ziv, G., Kumar, V., Marqués, M., Schuhmacher, M., Acuña,
501 V., 2016. Analysis of the uncertainty in the monetary valuation of ecosystem services — A case
502 study at the river basin scale. *Sci. Total Environ.* 543, Part, 683–690.

doi:<http://dx.doi.org/10.1016/j.scitotenv.2015.11.066>

Brauman, K.A., 2015. Hydrologic ecosystem services: linking ecohydrologic processes to human well-being in water research and watershed management. *Wiley Interdiscip. Rev. Water* 2, 345–358. doi:[10.1002/wat2.1081](https://doi.org/10.1002/wat2.1081)

Brauman, K.A., Daily, G.C., Duarte, T.K., Mooney, H.A., 2007. The Nature and Value of Ecosystem Services: An Overview Highlighting Hydrologic Services, in: *Annual Reviews (Ed.), Annual Review of Environment and Resources*. pp. 67–98. doi:<http://dx.doi.org/10.1016/j.advwatres.2011.02.010>

Connor, J.D., Bryan, B.A., Nolan, M., Stock, F., Gao, L., Dunstall, S., Graham, P., Ernst, A., Newth, D., Grundy, M., Hatfield-Dodds, S., 2015. Modelling Australian land use competition and ecosystem services with food price feedbacks at high spatial resolution. *Environ. Model. Softw.* 69, 141–154. doi:<http://dx.doi.org/10.1016/j.envsoft.2015.03.015>

Chapra, S.C., Pelletier, G.J., Tao, H., 2005. QUAL2K: a modeling framework for simulating river and stream water quality. Version 2.02.

Davies, E.G.G., Simonovic, S.P., 2011. Global water resources modelling with an integrated model of the social-economic-environmental system. *Adv. Water Resour.* 34, 684–700. doi:<http://dx.doi.org/10.1016/j.advwatres.2011.02.010>

de Groot, R.S., Wilson, M.A., Boumans, R.M.J., 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecol. Econ.* 41, 393–408. doi:[http://dx.doi.org/10.1016/S0921-8009\(02\)00089-7](http://dx.doi.org/10.1016/S0921-8009(02)00089-7)

Dennedy-Frank, P.J., Muenich, R.L., Chaubey, I., Ziv, G., 2016. Comparing two tools for ecosystem service assessments regarding water resources decisions. *J. Environ. Manage.* 177, 331–340. doi:<http://dx.doi.org/10.1016/j.jenvman.2016.03.012>

Díaz, S., Demissew, S., Carabias, J., Joly, C., Lonsdale, M., Ash, N., Larigauderie, A., Adhikari, J.R., Arico, S., Báldi, A., Bartuska, A., Baste, I.A., Bilgin, A., Brondizio, E., Chan, K.M.A., Figueroa, V.E., Duraipappah, A., Fischer, M., Hill, R., Koetz, T., Leadley, P., Lyver, P., Mace, G.M., Martin-Lopez, B., Okumura, M., Pacheco, D., Pascual, U., Pérez, E.S., Reyers, B., Roth, E., Saito, O., Scholes, R.J., Sharma, N., Tallis, H., Thaman, R., Watson, R., Yahara, T., Hamid, Z.A., Akosim, C., Al-

531 Hafedh, Y., Allahverdiyev, R., Amankwah, E., Asah, S.T., Asfaw, Z., Bartus, G., Brooks, L.A.,
 532 Caillaux, J., Dalle, G., Darnaedi, D., Driver, A., Erpul, G., Escobar-Eyzaguirre, P., Failler, P.,
 533 Fouda, A.M.M., Fu, B., Gundimeda, H., Hashimoto, S., Homer, F., Lavorel, S., Lichtenstein, G.,
 534 Mala, W.A., Mandivenyi, W., Matczak, P., Mbizvo, C., Mehrdadi, M., Metzger, J.P., Mikissa, J.B.,
 535 Moller, H., Mooney, H.A., Mumby, P., Nagendra, H., Nesshover, C., Oteng-Yeboah, A.A., Pataki,
 536 G., Roué, M., Rubis, J., Schultz, M., Smith, P., Sumaila, R., Takeuchi, K., Thomas, S., Verma, M.,
 537 Yeo-Chang, Y., Zlatanova, D., 2015. The IPBES Conceptual Framework — connecting nature and
 538 people. *Curr. Opin. Environ. Sustain.* 14, 1–16.
 539 doi:http://dx.doi.org/10.1016/j.cosust.2014.11.002

540 Du, J., Qian, L., Rui, H., Zuo, T., Zheng, D., Xu, Y., Xu, C.Y., 2012. Assessing the effects of urbanization
 541 on annual runoff and flood events using an integrated hydrological modeling system for
 542 Qinhua River basin, China. *J. Hydrol.* 464–465, 127–139.
 543 doi:http://dx.doi.org/10.1016/j.jhydrol.2012.06.057

544 Duan, Q., Sorooshian, S., Gupta, V.K., 1994. Optimal use of the SCE-UA global optimization method
 545 for calibrating watershed models. *J. Hydrol.* 158, 265–284. doi:10.1016/0022-1694(94)90057-4

546 Dyck, S. (Ed.), 1990. *Integrated Planning and Management of Water Resources*. United Nations
 547 Education, Scientific and Cultural Organization.

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

551 Fu, B., Wang, Y.K., Xu, P., Yan, K., 2013. Mapping the flood mitigation services of ecosystems – A
 552 case study in the Upper Yangtze River Basin. *Ecol. Eng.* 52, 238–246.
 553 doi:http://dx.doi.org/10.1016/j.ecoleng.2012.11.008

554 Fu, B., Wang, Y.K., Xu, P., Yan, K., 2012. Modelling nutrient retention function of ecosystem – a case
 555 study in Baoping County, China. *Procedia Environ. Sci.* 13, 111–121.
 556 doi:http://dx.doi.org/10.1016/j.proenv.2012.01.011

557 Global Water Partnership, 2000. *Integrated Water Resources Management, Background report 4*.

558 Grafton, R.Q., Pittock, J., Davis, R., Williams, J., Fu, G., Warburton, M., Udall, B., McKenzie, R., Yu, X.,

559 Che, N., Connell, D., Jiang, Q., Kompas, T., Lynch, A., Norris, R., Possingham, H., Quiggin, J.,
560 2013. Global insights into water resources, climate change and governance. *Nat. Clim. Chang.*
561 3, 315–321.

562 Guo, Z., Xiao, X., Li, D., 2000. An assessment of ecosystem services: Water flow regulation and
563 hydroelectric power production. *Ecol. Appl.* 10, 925–936. doi:[http://dx.doi.org/10.1890/1051-0761\(2000\)010\[0925:AAOESW\]2.0.CO;2](http://dx.doi.org/10.1890/1051-0761(2000)010[0925:AAOESW]2.0.CO;2)

565 Guswa, A.J., Brauman, K.A., Brown, C., Hamel, P., Keeler, B.L., Sayre, S.S., 2014. Ecosystem services:
566 Challenges and opportunities for hydrologic modeling to support decision making. *Water*
567 *Resour. Res.* 50, 4535–4544. doi:<http://dx.doi.org/10.1002/2014WR015497>

568 Herrera, S., Gutiérrez, J.M., Ancell, R., Pons, M.R., Frías, M.D., Fernández, J., 2012. Development
569 and analysis of a 50-year high-resolution daily gridded precipitation dataset over Spain
570 (Spain02). *Int. J. Climatol.* 32, 74–85. doi:<http://dx.doi.org/10.1002/joc.2256>

571 Karabulut, A., Egoh, B.N., Lanzaova, D., Grizzetti, B., Bidoglio, G., Pagliero, L., Bouraoui, F., Aloe, A.,
572 Reynaud, A., Maes, J., Vandecasteele, I., Mubareka, S., 2016. Mapping water provisioning
573 services to support the ecosystem-water-food-energy nexus in the Danube river basin. *Ecosyst.*
574 *Serv.* 17, 278–292. doi:<http://dx.doi.org/10.1016/j.ecoser.2015.08.002>

575 Keeler, B.L., Polasky, S., Brauman, K.A., Johnson, K.A., Finlay, J.C., O'Neill, A., Kovacs, K., Dalzell, B.,
576 2012. Linking water quality and well-being for improved assessment and valuation of
577 ecosystem services. *Proc. Natl. Acad. Sci.* 109, 18619–18624.
578 doi:<http://dx.doi.org/10.1073/pnas.1215991109>

579 La Notte, A., Maes, J., Grizzetti, B., Bouraoui, F., Zulian, G., 2012. Spatially explicit monetary
580 valuation of water purification services in the Mediterranean bio-geographical region. *Int. J.*
581 *Biodivers. Sci. Ecosyst. Serv. Manag.* 8, 26–34.
582 doi:<http://dx.doi.org/10.1080/21513732.2011.645557>

583 Liqueste, C., Cid, N., Lanzaova, D., Grizzetti, B., Reynaud, A., 2016. Perspectives on the link between
584 ecosystem services and biodiversity: The assessment of the nursery function. *Ecol. Indic.* 63,
585 249–257. doi:<http://dx.doi.org/10.1016/j.ecolind.2015.11.058>

586 Liqueste, C., Maes, J., La Notte, A., Bidoglio, G., 2011. Securing water as a resource for society: an

ecosystem services perspective. *Ecohydrol. Hydrobiol.* 11, 247–259.
doi:<http://dx.doi.org/10.2478/v10104-011-0044-1>

Maes, J., Liqueste, C., Teller, A., Erhard, M., Paracchini, M.L., Barredo, J.I., Grizzetti, B., Cardoso, A.,
Somma, F., Petersen, J.-E., Meiner, A., Gelabert, E.R., Zal, N., Kristensen, P., Bastrup-Birk, A.,
Biala, K., Piroddi, C., Egoh, B., Degeorges, P., Fiorina, C., Santos-Martín, F., Naruševičius, V.,
Verboven, J., Pereira, H.M., Bengtsson, J., Gocheva, K., Marta-Pedroso, C., Snäll, T., Estreguil,
C., San-Miguel-Ayanz, J., Pérez-Soba, M., Grêt-Regamey, A., Lillebø, A.I., Malak, D.A., Condé, S.,
Moen, J., Czúcz, B., Drakou, E.G., Zulian, G., Lavalley, C., 2016. An indicator framework for
assessing ecosystem services in support of the EU Biodiversity Strategy to 2020. *Ecosyst. Serv.*
17, 14–23. doi:<http://dx.doi.org/10.1016/j.ecoser.2015.10.023>

Milhous, R.T., 1990. Reference Manual for Generation and Analysis of Habitat Time Series-Version
II.

Ministerio de Agricultura Alimentación y Medio Ambiente, 2012. Propuesta de Proyecto de Plan
Hidrológico Nacional. Anejo 5 - Demandas de agua.

Momblanch, A., Connor, J.D., Crossman, N.D., Paredes-Arquiola, J., Andreu, J., 2016. Using
ecosystem services to represent the environment in hydro-economic models. *J. Hydrol.* 538,
293–303. doi:[10.1016/j.jhydrol.2016.04.019](http://dx.doi.org/10.1016/j.jhydrol.2016.04.019)

Natural Capital Project, 2016. www.naturalcapitalproject.org [WWW Document].

Paredes-Arquiola, J., Andreu-Álvarez, J., Martín-Monerris, M., Solera, A., 2010. Water Quantity and
Quality Models Applied to the Júcar River Basin, Spain. *Water Resour. Manag.* 24, 2759–2779.
doi:<http://dx.doi.org/10.1007/s11269-010-9578-z>

Paredes-Arquiola, J., Solera, A., Andreu, J., Lerma, N., 2014a. Herramienta EvalHid para la
evaluación de recursos hídricos.

Paredes-Arquiola, J., Solera, A., Martínez-Capel, F., Momblanch, A., Andreu, J., 2014b. Integrating
water management, habitat modelling and water quality at the basin scale and environmental
flow assessment: case study of the Tormes River, Spain. *Hydrol. Sci. J.* 59, 878–889.
doi:<http://dx.doi.org/10.1080/02626667.2013.821573>

614 Pulido-Velazquez, M., Andreu, J., Sahuquillo, A., 2006. Economic Optimization of Conjunctive Use of
615 Surface Water and Groundwater at the Basin Scale. *J. Water Resour. Plan. Manag.* 132, 454–
616 467. doi:[http://dx.doi.org/10.1061/\(ASCE\)0733-9496\(2006\)132:6\(454\)](http://dx.doi.org/10.1061/(ASCE)0733-9496(2006)132:6(454))

617 Pulido-Velazquez, M., Andreu, J., Sahuquillo, A., Pulido-Velazquez, D., 2008. Hydro-economic river
618 basin modelling: The application of a holistic surface–groundwater model to assess
619 opportunity costs of water use in Spain. *Ecol. Econ.* 66, 51–65.
620 doi:<http://dx.doi.org/10.1016/j.ecolecon.2007.12.016>

621 Richter, B.D., Thomas, G.A., 2007. Restoring environmental flows by modifying dam operations.
622 *Ecol. Soc.* 12, 12.

623 Sample, J.E., Baber, I., Badger, R., 2016. A spatially distributed risk screening tool to assess climate
624 and land use change impacts on water-related ecosystem services. *Environ. Model. Softw.* 83,
625 12–26. doi:<http://dx.doi.org/10.1016/j.envsoft.2016.05.011>

626 Tallis, H.T., Ricketts, T., Guerry, A.D., Wood, S.A., Sharp, R., Nelson, E., Ennaanay, D., Wolny, S.,
627 Olwero, N., Vigerstol, K., Pennington, D., Mendoza, G., Aukema, J., Foster, J., Forrest, J.,
628 Cameron, D., Arkema, K., Lonsdorf, E., Kennedy, C., Verutes, G., Kim, C.K., Guannel, G.,
629 Papenfus, M., Toft, J., Marsik, M., Bernhardt, J., Griffin, R., 2013. InVEST 2.5.4 User’s Guide.

630 Terrado, M., Acuña, V., Ennaanay, D., Tallis, H., Sabater, S., 2014. Impact of climate extremes on
631 hydrological ecosystem services in a heavily humanized Mediterranean basin. *Ecol. Indic.* 37,
632 199–209. doi:<http://dx.doi.org/10.1016/j.ecolind.2013.01.016>

633 Terrado, M., Momblanch, A., Bardina, M., Boithias, L., Munné, A., Sabater, S., Solera, A., Acuña, V.,
634 2016. Integrating ecosystem services in river basin management plans. *J. Appl. Ecol.* 53, 865–
635 875. doi:<http://dx.doi.org/10.1111/1365-2664.12613>

636 The Economics of Ecosystems & Biodiversity, 2010. The Economics of Ecosystems and Biodiversity:
637 Ecological and Economic Foundations. Earthscan.

638 Vigerstol, K.L., Aukema, J.E., 2011. A comparison of tools for modeling freshwater ecosystem
639 services. *J. Environ. Manage.* 92, 2403–2409.
640 doi:<http://dx.doi.org/10.1016/j.jenvman.2011.06.040>

641 Villa, F., Bagstad, K.J., Voigt, B., Johnson, G.W., Portela, R., Honzák, M., Batker, D., 2014. A
 642 Methodology for Adaptable and Robust Ecosystem Services Assessment. PLoS One 9, e91001.
 643 doi:http://dx.doi.org/10.1371/journal.pone.0091001

644 Wagner, P.D., Kumar, S., Schneider, K., 2013. An assessment of land use change impacts on the
 645 water resources of the Mula and Mutha Rivers catchment upstream of Pune, India. Hydrol.
 646 Earth Syst. Sci. 17, 2233–2246. doi:10.5194/hess-17-2233-2013

647 Watson, K.B., Ricketts, T., Galford, G., Polasky, S., O’Niel-Dunne, J., 2016. Quantifying flood
 648 mitigation services: The economic value of Otter Creek wetlands and floodplains to
 649 Middlebury, VT. Ecol. Econ. 130, 16–24. doi:http://dx.doi.org/10.1016/j.ecolecon.2016.05.015

650 Yang, X., Ren, L., Singh, V.P., Liu, X., Yuan, F., Jiang, S., Yong, B., 2012. Impacts of land use and land
 651 cover changes on evapotranspiration and runoff at Shalamulun River watershed, China.
 652 Hydrol. Res. 43. doi:10.2166/nh.2011.120

653 Yates, D., Purkey, D., Seiber, J., Huber-Lee, A., Galbraith, H., 2005. WEAP21 - A Demand, Priority,
 654 and Preference Driven Water Planning Model. Part 1. Water Int. 30.
 655 doi:http://dx.doi.org/10.1080/02508060508691894

656



DSPACE

<https://dspace.org/>

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

Momblanch Benavent, Andrea; Paredes-Arquiola, J.; Andreu, Joaquín

2017-04-10

Elsevier

<http://dx.doi.org/10.1016/j.envsoft.2017.03.033>

Momblanch A, Paredes-Arquiola J, Andreu J, Improved modelling of the freshwater provisioning ecosystem service in water scarce river basins, Environmental Modelling and Software, Volume 94, August 2017, Pages 87 – 99.