Working Document 14/01

Modeling water scarcity and droughts to analyze water policies in the Jucar Basin (Spain)

Mohamed Taher Kahil, Ariel Dinar and Jose Albiac

Department of Agricultural Economics
Agrifood Research and Technology Center (CITA-Government of Aragon)

This Working Document draws from the first chapter of the Ph.D. dissertation by Mohamed Taher Kahil "Water management policies under scarcity and climate change: Analysis of economic instruments, institutions and collective action", Department of Economic Analysis of the University of Zaragoza, Spain. The research has been supported by projects BIL13MA072 from MAPFRE and project INIA RTA2010-00109-C04 from the Spanish Ministry of Economy and Competitiveness. The ideas and opinions in this document are responsibility of the authors and not those of the institutions supporting the research.

To obtain reprints of the Working Document contact:

Jose Albiac
Unidad Economia Agraria
CITA-DGA
Avenida de Montañana 930
50.059 Zaragoza
Spain

email: maella@unizar.es
Phone: +34 976716351
Fax: +34 976716335

Summary

Growing water extractions combined with emerging demands for environment protection increase competition for scarce water resources worldwide, especially in arid and semiarid regions. In those regions, climate change is projected to exacerbate water scarcity and increase the recurrence and intensity of droughts. These circumstances call for methodologies that can support the design of sustainable water management. This paper presents a hydro-economic model that links a reduced form hydrological component, with economic and environmental components. The model is applied to an arid and semiarid basin in Southeastern Spain to analyze the effects of droughts and to assess alternative adaptation policies: institutional cooperation, water markets and water pricing. Results indicate that drought events have large impacts on social welfare, with the main adjustments sustained by irrigation and the environment. The water market policy seems to be a suitable option, although the environmental effects may weaken its advantages for society. The water pricing policy is the worst option in terms of private and environmental benefits. Because of their very large profit losses, farmers will oppose strongly water pricing, and the measure would become politically unfeasible. The current water management approach in Spain, based on stakeholders' cooperation, achieves almost the same economic outcomes and better environmental outcomes compared to a pure water market. These findings call for a reconsideration of the current management in basins around the world. The paper illustrates the potential of hydro-economic modeling for integrating the multiple dimensions of water resources, becoming a valuable tool in the advancement of sustainable water management policies.

Keywords: hydro-economic modeling, droughts, climate change, stakeholders' cooperation, water markets, water pricing, environmental benefits

1. Introduction

The pressure on water resources has been mounting worldwide with water scarcity becoming a widespread problem in most arid and semiarid regions around the world. Global water extractions have increased more than six-fold in the last century, which is more than twice the rate of human population growth. The huge exploitation of water resources has resulted in 35 percent of the world population living in regions with severe water scarcity. Furthermore, about 65 percent of global river flows and aquatic ecosystems are under moderate to high threats of degradation [*Alcamo et al.*, 2000; *Vörösmarty et al.*, 2010].

Projected future climate change impacts would further exacerbate the current situation of water scarcity in arid and semiarid regions. These regions would likely experience more severe and frequent droughts, making future water management even more difficult [*IPCC*, 2007]. The impacts of droughts in arid and semiarid regions can be substantial because they add on to the existing water scarcity situation. This is the case of recent droughts in Australia, the western United States, southern Europe, and Africa.

Severe droughts could have large impacts on agriculture, domestic and industrial users, tourism, and on ecosystems. Costs of drought damages seem to be considerable, and have been estimated to range from $2 to $6 billion per year in the United States [*FEMA*, 1995; *NOAA*, 2008], and around 3 billion € per year in the European Union [*EC*, 2007]. These costs represent between 0.05 and 0.1 percent of the gross domestic product (GDP), although the costs of drought could be exceptionally higher some years. Losses in the Murray-Darling basin (Australia) during 2009 were 20 percent of the value of irrigated agriculture, representing about 1 percent of GDP [*Kirby et al.*, 2014].

The scale and costs of the global growing overdraft of water resources indicates that water mismanagement is quite common, and that sustainable management of basins is a complex and difficult task. These difficulties call for the development of methodologies that allow a better understanding of water management issues within the contexts of scarcity, drought, and climate change. Integrated hydro-economic modeling is a potential methodology for implementing comprehensive river basin scale analysis to support the design of sustainable water management policies.

This methodology to model river basin interactions has been previously used in several studies, such as *Booker and Young* [1995], *McKinney et al.* [1999], *Cai et al.* [2003], *Booker et al.* [2005], *Lund et al.* [2006], *Pulido et al.* [2008], and *Ward* [2009]. The present paper suggests a prototype river basin hydro-economic model that links a reduced form hydrological component, with a regional economic optimization component and an environmental component. The reduced form hydrological component is calibrated to observed water allocations in normal and drought years using a regression approach. This new simple approach calibrates adequately the hydrological component and captures the basin response flexibility to various water availability levels, when detailed hydrological information is not available (which is the case in many basins worldwide). The regional economic component includes a detailed farm-level optimization model and an urban social surplus model, which differs from the usually used piece-wise linear or quadratic equations, exogenously generated, relating water use to economic benefits. The environmental component estimates the benefits that environmental amenities provide to society in a way that makes them comparable with the benefits derived from other uses, which is a challenging task in hydro-economic modeling.

The integrated model simulates demand nodes' behavior under different drought scenarios and policy intervention alternatives (institutional, agriculture-urban water market, and environmental water market policies). The linkage between model components allows a rigorous evaluation of drought impacts under the different policy settings: allocation among sectors, spatial distribution, use of surface and groundwater, land use decisions, and private and social benefits and costs of water utilization. The hydro-economic model is empirically tested in a semiarid basin in Southeastern Spain. The empirical application provides a valuable illustration of the development procedure of hydro-economic modeling, data requirements and calibration processes, as well as its use for comprehensive river basin climate and policy impact assessment.

The contributions of this paper relative to prior literature are both methodological and empirical ones, and the insights could be generalized for addressing the current mismanagement pervading the main basins in arid and semiarid regions around the world. The methodology combines three key elements partially tackled in previous hydro-economic modeling: a simplified hydrology circumventing full hydrological knowledge, a regional model including all economic sectors, and an explicit benefit function of basin ecosystems. This approach could be easily applied to most basins around the world.

Empirically, the results show the advantages of stakeholders' cooperation for water management. This is the institutional approach being implemented in Spain to address water scarcity, where stakeholders themselves participate in the design of management rules and implementation of enforcement mechanisms. The results show that this institutional approach achieves almost the same economic outcomes and better environmental outcomes compared to a pure water market policy (Pareto-efficient solution). These findings call for a reconsideration of the current water institutions and policies in many arid and semiarid basins, based on command and control instruments or else on pure economic instruments, such as water markets or water pricing. These instruments, that disregard stakeholders' role, have failed in reducing water scarcity and protecting ecosystems because they lack both legitimacy among stakeholders, and knowledge of local conditions. This empirical finding is an important policy issue for basins around the world, suggesting that collective action seems to be a key ingredient to move towards a more sustainable water management.

2. Modeling framework

The hydro-economic river basin model integrates hydrologic, economic, institutional, and environmental variables, and involves the main users in the basin, including irrigation districts, urban centers, and aquatic ecosystem requirements. The model is used to simulate various drought scenarios, and to assess the scope of possibilities to improve the environmental and economic outcomes of the basin under those drought scenarios.

Hydro-economic modeling is a powerful tool to analyze water scarcity, drought, and climate change issues. These models represent all major spatially distributed hydrologic and engineering parts of the studied river basin. Moreover, hydro-economic models allow capturing the effects of the interactions between the hydrologic and the economic systems, ensuring that the optimal economic results take into account the spatial distribution of water resources. The spatial location of water users, such as irrigation districts and households with respect to the river stream determines largely the magnitude of the impacts of any allocation decision and policy intervention to cope with water scarcity [*Harou et al.*, 2009; *Maneta et al.*, 2009].

However, developing the hydrologic part of the model is a time-consuming and complex task that involves detailed hydrologic knowledge and highly-disaggregated biophysical information that may not be available, requiring advanced modeling abilities that could represent the complex hydrological relationships. Moreover, hydrologic and economic models usually have different resolution techniques, and spatial and temporal scales, which further complicate their linkage [*Harou et al.*, 2009]. An alternative approach is to use aggregated historical data provided by water authorities, together with simulated data and network topology from existing hydrologic models. This method is a quick and credible way to build a reduced form hydrological model of the studied river basin [*Cai et al.*, 2003].

The reduced form hydrological model is a node-link network, in which nodes represent physical units impacting the stream system, and links represent the connection between these units. The nodes that could be included in the network are classified into two types: supply nodes, such as rivers, reservoirs, and aquifers; and demand nodes, such as irrigation districts, households, and aquatic ecosystems. The links could be rivers or canals (See below the representation of the Jucar model in figure 3).

The flows of water are routed between nodes using basic hydrologic concepts, such as mass balance and river flow continuity equations. The mass balance principle could be applied for surface flow, reservoir, and aquifer levels. The model is initially constrained by a known volume of water availability into the basin, and this volume can be varied depending on climate scenarios. Boundary conditions in the form of lower and upper bound constraints, such as minimum volume of water stored in reservoirs and maximum reservoirs and aquifers depletion, could be incorporated anywhere in the network. Institutional constraints could be added to the network to characterize the basin's allocation rules. River basin authorities worldwide have developed numerous institutional rules to allocate water among uses for political, legal, or environmental reasons. Examples include water rights, water sharing arrangements, and minimum environmental flows of river reaches. These constraints typically limit the choice of the hydro-economic model to optimally allocate water among uses [*Ward*, 2009].

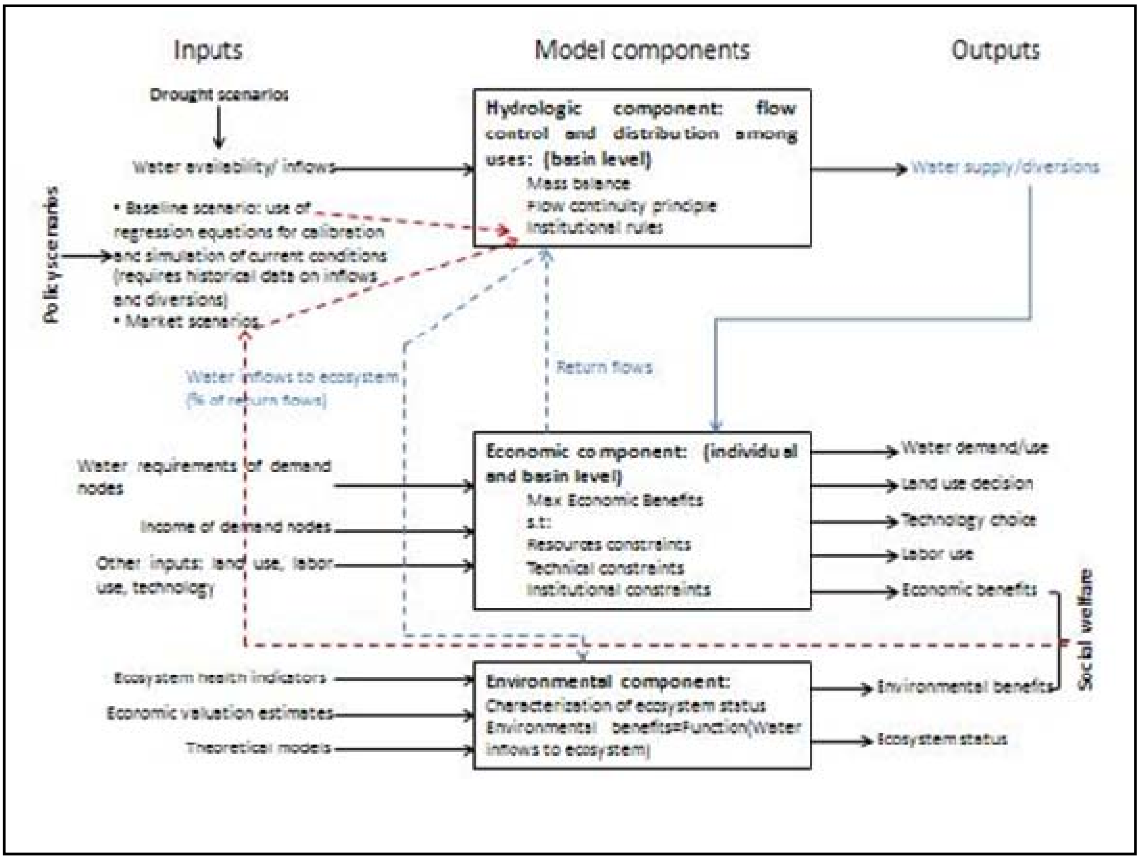
The development of the reduced form hydrological model requires accurate information on the geographical location of both supply and demand nodes, and the links and interactions between them (such as surface water diversion, groundwater extractions, return flows, wastewater discharge, reuse), and physical characterization of the nodes. Additionally, the model development needs information on water inflows (available runoff) time series measured at the considered headwater stream gauges, time series data on water use of demand nodes, streamflow time series data measured or estimated at selected river gauges, and infrastructure features at each node, including facility capacities, losses, and evaporation.

The reduced form hydrological model allows controlling the flows of water in each node and estimating the distribution of the available water among users under each climate condition. The model is calibrated so that predicted allocations to users in both normal and drought periods match historical water allocations in those periods. The calibration process involves defining time series data on streamflows at the considered stream diversion gauges, and the diversion of water for the demand nodes from those gauges during normal flow and drought years. In this paper, a regression approach modeling the relationship between water availability and diversion at each node has been used to calibrate the reduced form hydrological model. The calibration of the model may pose difficulties derived from the unobserved variables involved in the water allocation decisions, and the uncertainty linked to water use data. *Letcher et al.* [2007] suggest that integrated models should not be developed for prediction purposes, but to support the understanding of basin responses to changes, such as climate or policy changes.

The reduced form hydrological model, once calibrated, is incorporated into an economic framework. The linkage between the hydrologic and economic components requires adding several relationships that allow transferring information and feedback from one model component to the other. The economic benefits from water use in the irrigation sector are jointly determined using calibrated mathematical programming models that search for the optimal behavior of irrigation demand nodes subject to a set of technical and resource constraints. Alternatively, empirically estimated benefit functions, using econometric models that rely on the observed behavior of irrigation demand nodes could be used. Generally, calibrated mathematical programming models are computationally intensive, while econometric models are data intensive. The required data for econometric models is usually not available at a scale suitable for regional analysis, and they are less suitable for changing economic and biophysical conditions [*Young and Loomis*, 2014].

The economic benefits from urban water use are often found by measuring the social surplus derived from inverse water demand functions estimated using econometric techniques. Demand functions relate water use to the price of water and other explanatory variables such as income, climate, and household structure [*Young and Loomis*, 2014]. Environmental benefits provided by aquatic ecosystems could be modeled by developing ecological response models of those ecosystems and using existing economic valuation studies [*Keeler et al.*, 2012]. Otherwise, environmental

Figure 1. Modeling framework.



water uses may be represented with minimum-flow constraints if environmental valuation studies and ecosystem health indicators are unavailable.

The integrated hydro-economic model could then be used to simulate the effects of various drought scenarios on water uses in the studied river basin under the current institutional and policy setting predefined by the modeler. The procedure is as follows: (1) the calibrated reduced form hydrological model predicts water flows in each node and endogenously provides water availability constraints (supply) to the economic and environmental models, and (2) the economic and environmental models simultaneously determine water demand in each node to maximize nodes' economic benefits from water use. Different policy constraints could be added to the underlying framework or some existing constraints could be relaxed to investigate alternative allocation rules, institutional arrangements and policy interventions.

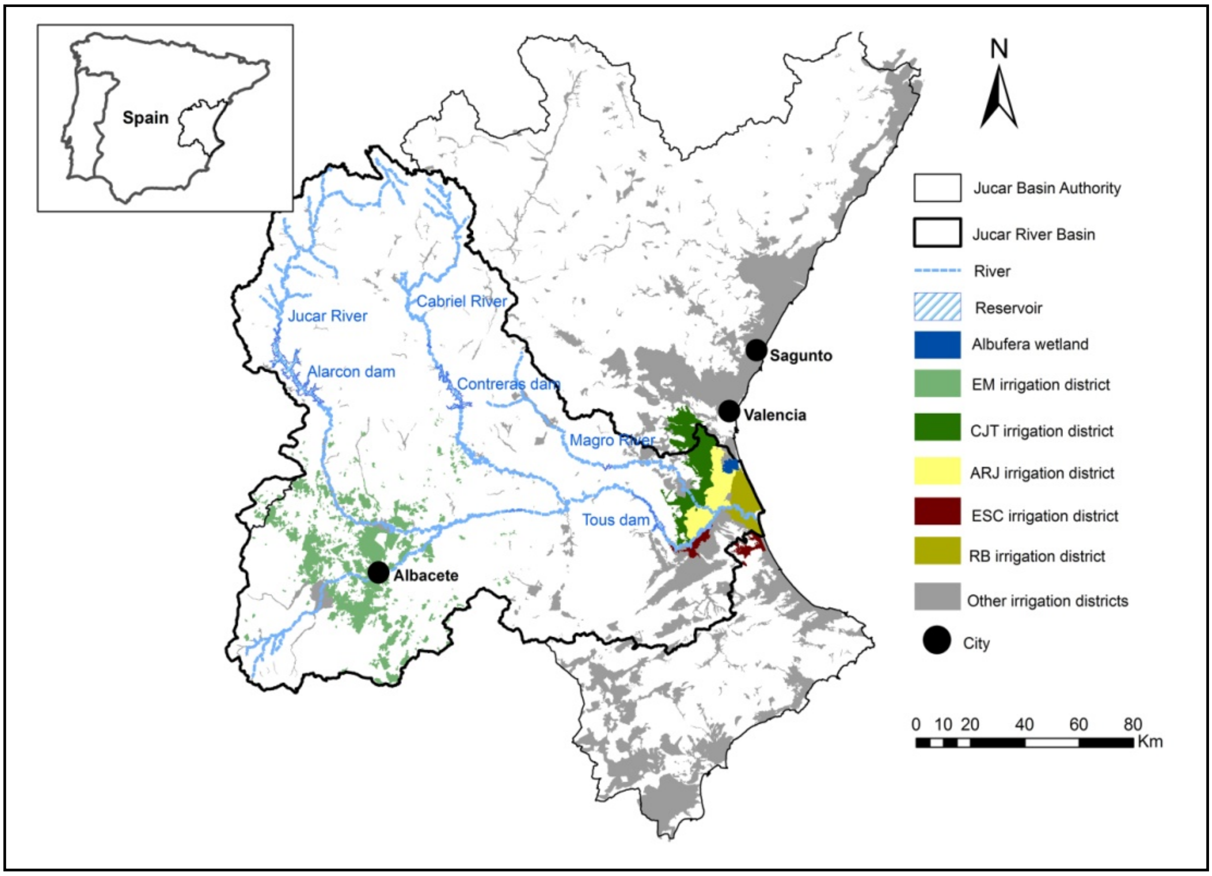
The modeling framework described in this section is summarized in figure 1 and it is applied to the drought management problem in a semiarid basin in Southeastern Spain, the Jucar River Basin. The next section provides background information on the basin, and the following sections present the design and calibration of the reduced form hydrological model and that of the economic models to the conditions in the Jucar River Basin.

3. The Jucar River Basin: Background information

The approach to water management in Spain is institutional and relies on the river basin authorities. The rationale behind that approach is the different types of goods and services provided by water, which can be classified as private goods, common pool resources, or public goods. Treated drinkable water in urban networks is close to a private good, irrigation water from surface watercourses and aquifers is close to a common pool resource, while water sustaining ecosystems comes close to a public good [*Booker et al.*, 2012]. The common pool and public good characteristics of water is a good reason for the institutional approach based on basin authorities achieving the collective action of stakeholders.

The basin authorities in Spain are responsible for water management, water allocation and water public domain, planning and waterworks. The special characteristic of this institutional approach is the key role played by stakeholders in basin authorities. Stakeholders are inside basin authorities taking decisions in the basin governing bodies and in local watershed boards, and they are involved at all levels of decision making: planning, financing, waterworks, measures design, enforcement, and water

Figure 2. Map of the Jucar River Basin.



management. The management of water is decentralized, with the basin authorities in charge of water allocation, and water user associations in charge of secondary infrastructure, water usage, operation and maintenance, investments, and cost recovery. The main advantage of this institutional setting is that stakeholders cooperate in the design and enforcement of decisions, rules and regulations, and therefore the implementation and enforcement processes are carried on smoothly.

The Jucar River Basin (henceforth JRB) is located in the regions of Valencia and Castilla La Mancha in Southeastern Spain (Figure 2). It extends over 22,300 Km^2 and covers the area drained by the Jucar River and its tributaries, mainly the Magro and the Cabriel Rivers. The basin has an irregular Mediterranean hydrology, characterized by recurrent drought spells and normal years with dry summers.

The basin includes 13 reservoirs, the most important of which are the Alarcon, Contreras and Tous dams. There are two major water distribution canals: the Acequia Real canal, which conveys water from the Tous dam to the traditional irrigation districts in the lower Jucar, and the Jucar-Turia canal, which transfers water from the Tous dam to irrigation districts located in the bordering Turia River Basin.

At present, renewable water resources in the JRB are nearly 1,700 Mm^3, of which 930 are surface water and 770 are groundwater resources. Water extractions are 1,680 Mm^3, very close to renewable resources, making the JRB an almost closed water system. Extractions for irrigated agriculture are nearly 1,400 Mm^3. Urban and industrial extractions total 270 Mm^3, which supply households, industries, and services of more than one million inhabitants, located mostly in the cities of Valencia, Sagunto and Albacete.

The irrigated area extends over 190,000 ha, and the main crops grown are rice, wheat, barley, garlic, grapes, and citrus. There are three major irrigation areas, the Eastern La Mancha irrigation area (henceforth EM) is located in the upper Jucar, the traditional irrigation districts of Acequia Real del Jucar (henceforth ARJ), Escalona y Carcagente (henceforth ESC), and Ribera Baja (henceforth RB) are in the lower Jucar, and the irrigation area of the Canal Jucar-Turia (henceforth CJT) is located in the bordering Turia River Basin.

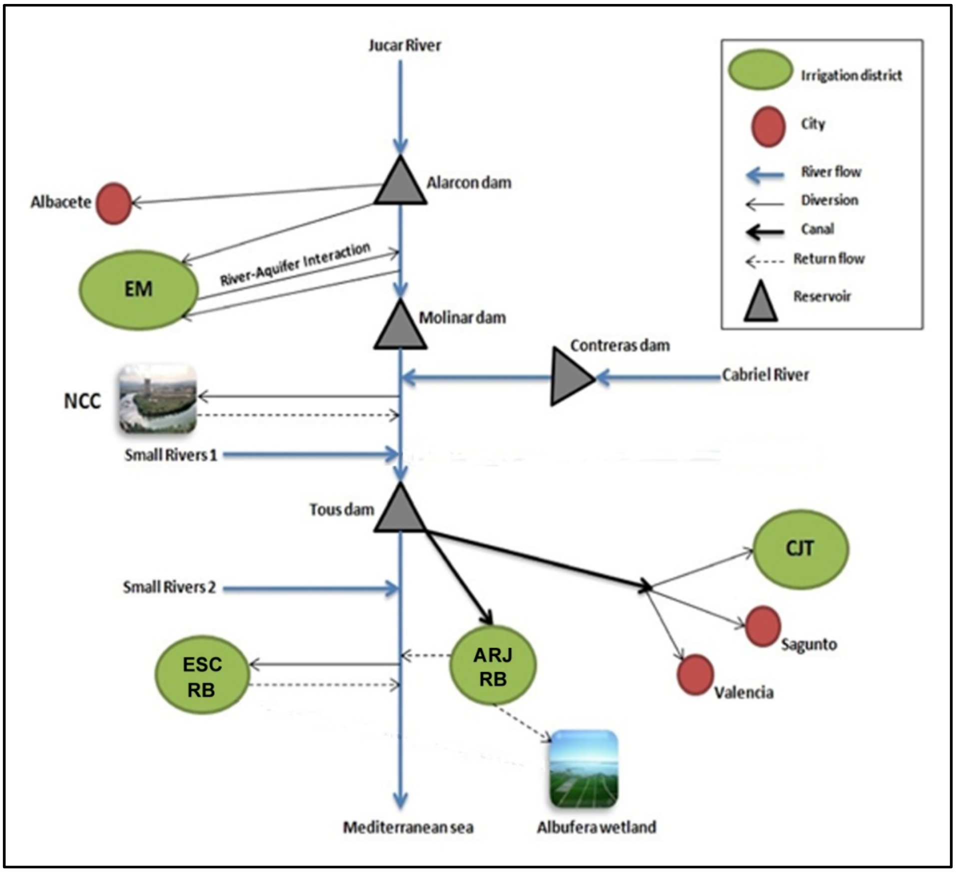
The expansion of water extractions and the severe drought spells in recent decades have triggered considerable negative environmental and economic impacts in the basin [*CHJ*, 2009]. The growth of water extractions has been driven especially by groundwater irrigation from the EM aquifer. The aquifer water table has dropped about 80 m in some areas, resulting in large storage depletion, fluctuating around 2,500 Mm^3.

The aquifer is linked to the Jucar River stream, and it fed the Jucar River with about 150 Mm^3/year in the 1980s. Due to the depletion, the aquifer is at present draining the water flow of the upper Jucar rather than feeding it, at an average of 70 Mm^3/year during 2001–2005 [*Sanz et al.*, 2011].

Environmental flows are dwindling in many parts of the basin, resulting in serious damages to water-dependent ecosystems. The environmental flow in the final tract of the Jucar River is below 1 m^3/s, which is very low compared with the other two major rivers in the region, the Ebro and Segura Rivers. In addition, there have been negative impacts on the downstream water users. For instance, the water available to the ARJ district has been reduced from 700 to 200 Mm^3 in the last 40 years. Consequently, the dwindling return flows from the irrigation districts have caused serious environmental problems to the Albufera wetland, the main aquatic ecosystem in the JRB, which is mostly fed by these return flows [*Garcia-Molla et al.*, 2013].

The Albufera wetland is a freshwater lagoon with an area covering 2,430 ha, supporting very rich aquatic ecosystems. The Albufera is a RAMSAR site and a special protected area for birds. The Albufera receives water from the return flows of the ARJ and RB irrigation districts. Other flows originate from the Turia River Basin, and from discharges of untreated and treated urban and industrial wastewaters in the adjacent municipalities. At present, the wetland suffers from the reduction of inflows originating from the Jucar River and the degradation of water quality. The Jucar River flows play an important role in improving the quality of urban and industrial wastewater discharges to the wetland and in meeting its minimum water requirements. Water inflows reduction and quality degradation has caused severe damages to the Albufera wetland, triggering the decline of fish populations and recreation services [*Sanchis*, 2011].

Figure 3. JRB network.



4. The model components

The hydro-economic model includes three components: (1) a reduced form hydrological model, (2) a regional economic model, and (3) an environmental benefit model. The features of each model and the estimation procedure used for its coefficients are described below (See Kahil et al. [2014] for further details).

4.1. Reduced form hydrological model

The model is applied, using data from the Jucar basin authority [*CHJ*, 2009]. The model is calibrated to water allocations in both normal and drought periods, taking into account the response of the basin authority to three consecutive years in the last drought period from 2006 to 2008. Figure 3 presents the hydrological network of the basin, including the most important infrastructures, and water supply and demand nodes.

The reduced form hydrological model of the JRB is based on the principles of water mass balance and continuity of river flow, which determine the volume of water availability that can be used for economic activities after considering the environmental restrictions. The mathematical formulation of the reduced form model is as follows:

$$Wout_d = Win_d - Wloss_d - Div_d^{IR} - Div_d^{URB} \tag{1}$$

$$Win_{d+1} = Wout_d + r_d^{IR} \cdot (Div_d^{IR}) + r_d^{URB} \cdot (Div_d^{URB}) + RO_{d+1} \tag{2}$$

$$Wout_d \geq E_d^{min} \tag{3}$$

The mass balance equation (1) determines the water outflow $Wout_d$ from a river reach *d*, which is equal to water inflow Win_d minus the loss of water $Wloss_d$ (including evaporation, seepage to aquifers and any other loss) and the diversions for irrigation Div_d^{IR}, and urban and industrial uses Div_d^{URB}. The continuity equation (2) guarantees the continuity of river flow, where the water inflow to the next river reach Win_{d+1} is the sum of outflow from upstream river reach $Wout_d$, the return flows from previous irrigation districts $[r_d^{IR} \cdot (Div_d^{IR})]$, the return flows from the cities $[r_d^{URB} \cdot (Div_d^{URB})]$, and runoff entering that river reach from tributaries, RO_{d+1}. Equation (3) states that the water outflow $Wout_d$ from a river reach *d* must be greater than or equal to the minimum environmental flow E_d^{min} in that river reach.

Water diversions for irrigation districts Div_d^{IR} and for urban and industrial uses Div_d^{URB}, and minimum environmental flows E_d^{min}, are governed by a set of allocation rules defined in the JRB's regulations, which are implemented by the basin authority in response to climate conditions and reservoir storage. The hydrological plan of the JRB defines surface water allocations in the basin following the historical water rights and the access to groundwater resources. The Alarcon agreement of 2001 transferred the ownership of the Alarcon dam from farmers in the lower Jucar with seniority rights to the public administration, in exchange for guarantees on water rights and water use priority to these traditional districts. The agreement establishes that during drought situations, selected users could continue extracting surface water but they have to pay compensation to the traditional irrigation districts that are reducing surface extractions. Additionally, these traditional districts get a special authorization to substitute surface water for groundwater during drought, and the compensation covers the costs of groundwater pumping.

The JRB drought plan, approved in 2007, includes an integrated system of hydrological indicators that are used to declare the state of alert or full drought. Drought events trigger progressively stronger measures as the drought situation worsens. The drought plan allocates water following the priority rules that guarantee the provision of urban, industrial and environmental demand, while giving low priority to irrigation [*CHJ*, 2007]. The draft of the upcoming hydrological plan of the JRB proposes minimum environmental flows for the different reaches of the Jucar River, based on technical studies that evaluate ecosystem needs for each reach [*CHJ*, 2009].

Water diversions for the different uses under the current institutional setting have been approximated by regression equations. These equations model the relationship between water diversion for each demand node (Div_d^{IR} or Div_d^{URB}, as dependent variables) and the net water inflow to the corresponding river reach (Win_d, as an explanatory variable). These relationships have been calculated using data on water diversions and water inflows in each diversion node for a normal flow year and for each year in the drought period (2006, 2007, and 2008). The advantage of using the regression approach instead of fixed allocation coefficients is that it captures implicitly the flexibility of the basin authority's response to drought including water allocation rules and reservoir operation regimes. The distinctive feature of the current management (baseline scenario) in the JRB is the institutional approach to water management, based on river basin authorities that organize the collective action of stakeholders. This approach is based on negotiated arrangements and stakeholders' cooperation. The water allocations in the baseline scenario are the result of this collective action process. These allocations are captured in the model through the use of the regression equations. When water market scenarios are simulated, the regression equations are removed from the model, and market-based (equi-marginal principle) water allocations are driven by the optimization of economic benefits.

Information on groundwater extractions by demand node has been incorporated exogenously into the reduced form hydrological model to cover the demand of each node [*CHJ*, 2009]. It is assumed that groundwater use in the EM irrigation district decreases as drought severity intensifies, based on the observed cooperative behavior of farmers in the last two decades. This behavior is driven by the pressures of the basin authority with the political influence of the downstream stakeholders, calling for the control of extractions and threatening farmers by not issuing water rights [*Sanz et al.*, 2011; *Esteban and Albiac*, 2012]. Increases in groundwater extractions in certain irrigation districts are allowed by the basin authority during drought periods within the framework of the Alarcon agreement. These additional extractions are restricted in the model based on past maximum pumping levels [*IGME*, 2009]. In this paper, groundwater dynamics and pumping costs are held constant because of the short run nature of the model. Furthermore, the major groundwater extractions in the JRB are those of the EM aquifer, which is the largest aquifer system in Spain. Any changes in its water table level require a very long period of time.

The interaction between the Jucar River and EM aquifer has been approximated by a linear regression equation covering the period 1984 to 2004. The dependent variable is the discharge Q from aquifer to river, and the explanatory variable is groundwater pumping W_{GW}. This approximation follows the results by *Sanz et al.* [2011] indicating that there is a linear relationship between the Jucar River depletion and groundwater extraction in the EM aquifer. *Sanz et al.* [2011] find that although groundwater extractions increased considerably from 1980s, the depletion of the aquifer has been lower than expected because of the aquifer recharge coming from the Jucar River. Only a contemporary (one period) river-aquifer interaction is included in the reduced form hydrological model, given the short run or static nature of the analysis.

4.2. Regional economic model

The regional economic model accounts for the decision processes made by irrigation users in the five major irrigation districts (EM, CJT, ARJ, ESC, and RB) and by urban users in the three main cities (Valencia, Albacete, and Sagunto).

A farm-level model has been developed for each irrigation district, which maximizes farmers' private benefits of the chosen crop mix subject to technical and resource constraints. A Leontief production function technology is assumed with fixed input and output prices, in which farmers are price takers. The optimization problem is given by the following formulation:

$$Max\ B_k^{IR} = \sum_{ij} C'_{ijk} \cdot X_{ijk} \qquad (4)$$

subject to

$$\sum_i X_{ijk} \leq Tland_{kj}\ ;\ j = flood, sprinkler, drip \qquad (5)$$

$$\sum_{ij} W_{ijk} \cdot X_{ijk} \leq Twater_k \qquad (6)$$

$$\sum_{ij} L_{ijk} \cdot X_{ijk} \leq Tlabor_k \qquad (7)$$

$$X_{ijk} = \sum_n \alpha_n \cdot X_{ijkn} \; ; \; \sum_n \alpha_n = 1 \; ; \; \alpha_n \geq 0 \tag{8}$$

$$X_{ijk} \geq 0 \tag{9}$$

where B_k^{IR} is private benefit in irrigation district k and $C_{ijk}^{'}$ is net income per hectare of crop i using irrigation technology j. The decision variable in the optimization problem is X_{ijk}, the area of crop i under irrigation technology j. Crops are aggregated into three representative groups: cereals, vegetables, and fruit trees. Irrigation technologies are flood, sprinkler, and drip.

The land constraint (5) represents the irrigation area equipped with technology j in district k, $Tland_{kj}$. The water constraint (6) represents the water available in district k, $Twater_k$, which is the sum of surface water and groundwater extractions. Parameter W_{ijk} is gross water requirements per hectare of crop i with technology j. The water constraint level is the connecting variable between the economic optimization model of irrigation districts and the reduced form hydrological model. The labor constraint (7) represents labor availability in district k, $Tlabor_k$. Parameter L_{ijk} is labor requirements per hectare of crop i using technology j.

The aggregation constraint (8) forces crop production activities X_{ijk} to fall within a convex combination of historically observed crop mixes X_{ijkn}, where the index n indicates the number of the observed crop mixes. The aggregate supply response solution determines endogenously the weight variables α_n during the optimization process, because the optimal solution is the weighted sum of the corresponding crops mixes [*Önal and McCarl*, 1991]. Mathematical programming models have to account for the aggregation problem when performing an analysis at regional level, because farms are heterogeneous. The convex combination approach solves the aggregation problem using theoretical results from linear programming. Other procedures such as the representative farm approach and the positive mathematical programming make quite strong assumptions on farm responses.

Detailed information on the technical coefficients and parameters have been collected from field surveys, expert consultation, statistical reports, and reviewing the literature. This information covers crop yields and prices, subsidies, crop water and labor requirements and parcel irrigation efficiencies, water and production costs, land and labor availability, and groundwater extractions [*GV*, 2009; *GCLM*, 2009; *INE*, 2009; *MARM*, 2010]. The district models are calibrated for the year 2009 (a normal flow year), with observed crop area, water use, and net income of each irrigation district by crop group (Table 1).

For urban water uses, an economic surplus model has been developed for each city in the basin. The model maximizes social surplus given by the consumer and producer surplus from water use in each city, subject to several physical and institutional constraints. The optimization problem is:

$$Max\ B_u^{URB} = \left(a_{du} \cdot Q_{du} - \frac{1}{2} \cdot b_{du} \cdot Q_{du}^2 - a_{su} \cdot Q_{su} - \frac{1}{2} \cdot b_{su} \cdot Q_{su}^2\right) \tag{10}$$

subject to

$$Q_{du} - Q_{su} \leq 0 \tag{11}$$

$$Q_{du}\ ;\ Q_{su} \geq 0 \tag{12}$$

where B_u^{URB} is the consumer and producer surplus of city u. Variables Q_{du} and Q_{su} are water demand and supply by/to the city u, respectively. Parameters a_{du} and b_{du} are the intercept and slope of the inverse demand function, while parameters a_{su} and b_{su} are the intercept and slope of the water supply function. Equation (11) states that supply must be greater than or equal to demand. The quantity supplied, Q_{su}, is the connecting variable between urban use optimization models and the reduced form hydrological model. This research adapts the empirical water demand findings for Valencia, Albacete, and Sagunto from the study by *Collazos* [2004]. Urban water use decisions are simulated through the price mechanism, in which information on changed supplies is transmitted through price changes. Information on urban water prices and costs are taken from the Jucar basin authority reports [*CHJ*, 2009] (Table 1).

4.3. Environmental benefit model

The river basin model accounts for environmental benefits generated by the main aquatic ecosystem in the JRB, the Albufera wetland. Wetlands provide a wide range of services to society, including food production, groundwater recharge, nutrient cycling, carbon sequestration, habitat for valuable species, and recreational opportunities [*Woodward and Wui*, 2001]. Estimating wetland benefits in a way that makes them comparable with the benefits derived from other uses is helpful for the design of sustainable water management policies [*Turner et al.*, 2000].

The environmental benefit model developed here considers only water inflows to the Albufera wetland originating from irrigation return flows of the ARJ and RB irrigation districts. Inflows and benefits of the Albufera wetland are given by the following expressions:

$$E_{Albufera} = \alpha \cdot r_{ARJ}^{IR} \cdot \left(Div_{ARJ}^{IR}\right) + \beta \cdot r_{RB}^{IR} \cdot \left(Div_{RB}^{IR}\right) \qquad (13)$$

$$B_{Albufera} = \begin{cases} \delta_1 & if\ 0 \leq E_{Albufera} \leq E_1 \\ \delta_2 + \rho_2 \cdot E_{Albufera} & if\ E_1 < E_{Albufera} \leq E_2 \\ \delta_3 + \rho_3 \cdot E_{Albufera} & if\ E_2 < E_{Albufera} \leq E_3 \end{cases} \qquad (14)$$

where equation (13) determines the quantity of water flowing to the Albufera wetland from irrigation return flows, $E_{Albufera}$. Parameters α and β represent the shares of return flows that feed the wetland from the ARJ and RB irrigation districts, respectively. The products $[r_{ARJ}^{IR} \cdot (Div_{ARJ}^{IR})]$ and $[r_{RB}^{IR} \cdot (Div_{RB}^{IR})]$ are return flows from the ARJ and RB irrigation districts, respectively. Equation (14) represents economic environmental benefits, $B_{Albufera}$, from the services that the Albufera wetland provides to society. The economic environmental benefit function is assumed to be a piecewise linear function of water inflows, $E_{Albufera}$, to the wetland. This function expresses shifts in the ecosystem status when critical thresholds of environmental conditions (water inflows in this case) E_1 and E_2 are reached, while E_3 is the maximum observed inflow. This functional form is adapted from the study by *Scheffer et al.* [2001], indicating that ecosystems do not always respond smoothly to changes in environmental conditions, but they may switch abruptly to a contrasting alternative state when these conditions approach certain critical levels. $E_{Albufera}$ is the connecting variable between the environmental benefit model, the economic regional model, and the reduced form hydrological model.

Time series data of various ecosystem health indicators of the Albufera wetland have been collected including the quantity of water inflows, the number of water replenishments, *chlorophyll a* and phosphorus concentration, and salinity level, to calculate a unique health index of the wetland for each year of available data following the methodology developed by *Jorgensen et al.* [2010]. We assume that environmental benefits of the wetland are a function of its ecosystem health. With one year's worth of information about the economic value of the wetland, we extrapolate the economic value for each year of the available data using the health index of such year. Once the economic values are calculated for each year, the thresholds E_1 and E_2 in the health

Table 1. Parameters of the JRB model.

Parameters	Value	Unit
Total irrigated area	157,000	ha
Cereals area	70,650	ha
Vegetables area	21,980	ha
Fruit trees area	64,370	ha
Flood irrigation area	28,260	ha
Sprinkler irrigation area	58,090	ha
Drip irrigation area	70,650	ha
Average irrigation water price	0.05	€/m^3
Average urban water price	0.71	€/m^3
Inverse water demand functions for cities		
Intercept (a_{du})		
Valencia	6	€
Albacete	6	€
Sagunto	6	€
Slope (b_{du})		
Valencia	-0.06	€/Mm^3
Albacete	-0.3	€/Mm^3
Sagunto	-0.5	€/Mm^3
Benefit function of the Albufera from water inflows		
Intercept (δ_1)	33	10^6 €
First threshold of inflows to the Albufera (E_1)	51	Mm^3
Intercept (δ_2)	-214	10^6 €
Slope (ρ_2)	4.8	€/m^3
Second threshold of inflows to the Albufera (E_2)	78	Mm^3
Intercept (δ_3)	43	10^6 €
Slope (ρ_3)	1.8	€/m^3
Third threshold of inflows to the Albufera (E_3)	138	Mm^3
Economic value of the Albufera wetland	13,600	€/ha

index are determined, and the relationships between the environmental benefits and water inflows to the wetland are estimated.

The economic value of the Albufera wetland used to estimate the environmental benefit function is approximated, using the results from *Del Saz and Perez* [1999] on the recreation value of the Albufera wetland in the 1995, and other studies from the literature that estimate non-recreation values of wetlands [*Woodward and Wui*, 2001; *Brander et al.*, 2006]. The economic value used for estimating the environmental benefit function of the Albufera wetland, and the parameters estimates are presented in Table 1.

4.4. JRB optimization model

The JRB optimization model integrates the three components presented earlier. The model maximizes total basin benefits subject to the hydrological constraints and the constraints of the individual economic sector optimization models. The optimization problem for the whole river basin takes the following form:

$$Max \left(\sum_l B_l + B_{Albufera}\right) \quad \forall \, l = k, u \tag{15}$$

subject to the constraints in equations (1), (2), (3), (5), (6), (7), (8), (9), (11), (12), (13), and a set of constraints that defines the allocation of water among users depending on the policy intervention alternative that will be presented in section 6.1:

$$Div_d^l = f(Win_d) \quad \forall \, l, d \tag{16}$$

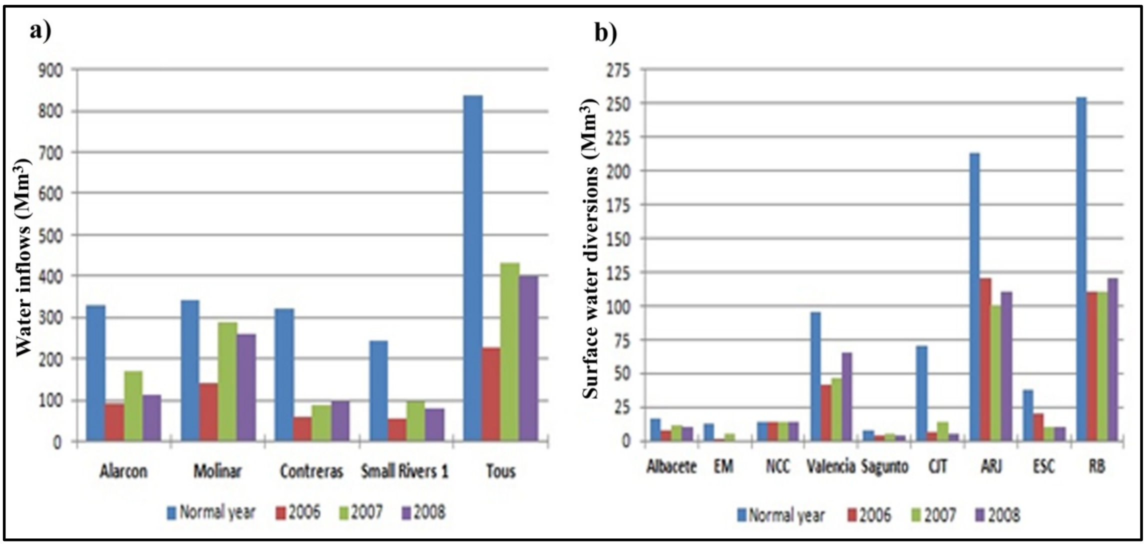
$$\sum_{ld} Div_d^l \leq \overline{W} \tag{17}$$

where B_l is the benefits of each demand node l and $B_{Albufera}$ is the environmental benefits provided by the Albufera wetland to society. Equations (16) and (17) are used to allocate water among users under the baseline scenario. Equations (16) ensures that water diversion, Div_d^l, for each demand node l located in a river reach d is a function, $f(.)$, of net water inflow to the corresponding river reach, Win_d. Equation (17) ensures that the sum of water diversions to all users, Div_d^l, does not exceed water available for the whole basin, $\overline{W}$. Under the water market scenarios, equation (16) is removed from the model and only equation (17) is used to allocate water among users.

5. Data sources and hydrological relationships

Information about water inflows to the main reservoirs and river reaches has been taken from the reports and modeling efforts of the Jucar basin authority. The annual reports provide historical data on gauged inflows in the basin, while the hydrological model of the JRB "AQUATOOL" provides additional information on the circulating flows in the basin [*CHJ*, 2002, 2012; *Deidda*, 2004; *Collazos*, 2004] (Figure 4).

Figure 4. Surface water inflows to the main reservoirs and river reaches (a) and diversions for the demand nodes (b) in the JRB.



Water diversions for irrigation have been calculated using detailed information on crop areas and water requirements, and irrigation technologies and efficiencies in each irrigation district [*INE*, 2009; *GV*, 2009; *GCLM*, 2009]. Water diversions for cities and industries have been taken from the Jucar basin authority [*CHJ*, 2002, 2009], where the water diversion to the nuclear power plant of Cofrentes (henceforth NCC) is always maintained at a fixed level (Figure 4).

Return flows have been calculated as the fraction of diverted water not used in crop evapotranspiration $[r_d^{IR} \cdot (Div_d^{IR})]$ and urban consumption $[r_d^{URB} \cdot (Div_d^{URB})]$. Most return flows originate from irrigation, with overall irrigation efficiency estimated at 60 percent, given the efficiency of farm plots and primary and secondary conveyance networks. Information about the distribution of return flows is taken from the reports of the basin authority [*CHJ*, 2009].

A good ecological status of the Albufera wetland is directly linked to the return flows from the ARJ and RB districts in the lower Jucar. Studies by the Jucar basin authority provide information on the amount and sources of water flows feeding the Albufera wetland during recent years [*CHJ*, 2009]. Following these studies, the

Table 2. Relationships between water diversions and inflows.

Demand nodes	Regression equations[*]	
Albacete[**]	$Div^{URB} = 5.2089 + 0.0358 \cdot \mathrm{Win}_{\mathrm{Alarcon}}$	(0.98)
EM irrigation district[**]	$Div^{IR} = -5.3319 + 0.0562 \cdot \mathrm{Win}_{\mathrm{Alarcon}}$	(0.98)
Jucar River-EM aquifer interaction[**]	$Q = 475.06 - 1.2214 \cdot \mathrm{W}_{\mathrm{GW}}$	(0.50)
Valencia[†]	$Div^{URB} = 21.806 + 0.086 \cdot \mathrm{Win}_{\mathrm{Tous}}$	(0.86)
Sagunto[***]	$Div^{URB} = 1.9201 + 0.007 \cdot \mathrm{Win}_{\mathrm{Tous}}$	(0.93)
CJT irrigation district[††]	$Div^{IR} = 22.44 - 0.1173 \cdot \mathrm{Win}_{\mathrm{Tous}} + 0.0002 \cdot \mathrm{Win}^2_{\mathrm{Tous}}$	(0.99)
ARJ irrigation district[†]	$Div^{IR} = 52.364 + 0.1761 \cdot \mathrm{Win}_{\mathrm{Tous}}$	(0.76)
ESC irrigation district[††]	$Div^{IR} = 1.344 + 0.0384 \cdot \mathrm{Win}_{\mathrm{Tous}}$	(0.57)
RB irrigation district[***]	$Div^{IR} = 31.25 + 0.1988 \cdot (\mathrm{Win}_{\mathrm{Tous}} + \mathrm{Win}_{\mathrm{SR1}} + \mathrm{r}^{\mathrm{IR}} \cdot \mathrm{Div}^{\mathrm{IR}})$	(0.91)

Note: $\mathrm{Win}_{\mathrm{Alarcon}}$= Water inflows to Alarcon dam; $\mathrm{Win}_{\mathrm{Tous}}$= Water inflows to Tous dam; $\mathrm{Win}_{\mathrm{SR1}}$= Water inflows from small rivers 1; $\mathrm{r}^{\mathrm{IR}} \cdot \mathrm{Div}^{\mathrm{IR}}$= Irrigation return flows from previous irrigation districts; W_{GW}= Groundwater pumping. [*] R^2 are in parenthesis; [**] Regression coefficients significant at $p<0.01$; [***] Regression coefficients significant at $p<0.05$; [†] Regression coefficients significant at $p<0.1$; [††] Regression coefficients significant at $p<0.2$.

Albufera receives 28 and 23 percent of the return flows from the ARJ and RB districts, respectively. These return flows distribution coefficients are held constant for all drought scenarios.

Table 2 presents the relationships between water diversions for demand nodes and water inflows to the diversion nodes, and also the Jucar River-EM aquifer relationship. For simplicity, all estimated relationships have been assumed linear, except in the case of the CJT irrigation district for which a quadratic specification seems more suitable. These equations are used to reproduce the observed water allocations to users under normal flow and drought years. After validation, they are used to simulate the allocation of water under the baseline scenario for the hypothetical future drought scenarios that will be presented in section 6.1.

The reduced form hydrological model is validated by comparing the simulated and observed values of water diversions in the demand nodes for normal flow and drought years. The robustness of the model results are tested using the coefficient of determination (R^2) and the Nash-Sutcliffe efficiency coefficient (NSE, ranges from 1 to $-\infty$) [*Krause et al.*, 2005]. The validation results verify the robustness of the reduced form hydrological model, because the values of R^2 range between 0.55 and 0.99, and the values of NSE range between 0.54 and 1. The outcomes are broadly consistent,

Table 3. Simulated (Sim) and Observed (Ob) water diversions (Mm^3).

Demand nodes	**Normal flow**		**2006**		**2007**		**2008**		**Statistical measures**	
	Sim	**Ob**	**Sim**	**Ob**	**Sim**	**Ob**	**Sim**	**Ob**	**R^2**	**NSE**
Albacete	17	17	8	8	11	11	9	10	0.99	0.98
EM	13	13	0	0.2	4	5	1	0	0.99	0.98
NCC	14	14	14	14	14	14	14	14	-	1
Valencia	94	95	41	42	59	47	56	66	0.86	0.86
Sagunto	8	8	3	4	5	5	5	4	0.84	0.81
CJT	64	70	6	7	9	14	7	5	0.99	0.98
ARJ	200	213	92	120	129	100	123	110	0.76	0.76
ESC	33	38	10	20	18	10	17	10	0.55	0.54
RB	243	254	87	110	136	110	126	120	0.91	0.91
Albufera	51	55	21	27	30	24	29	26	0.85	0.85
Total	**738**	**777**	**282**	**352**	**415**	**340**	**387**	**365**	**0.91**	**0.91**

indicating that the model reproduces adequately the hydrologic conditions. However, the differences between the simulated and observed outcomes are sizable for some demand nodes in both normal and drought years. These differences are probably explained by the limited number of observations in the data series used for the estimation (Table 3). A detailed description of the validation process can be found in *Kahil et al.* [2014].

6. Results and discussion

6.1. River basin model application and scenarios

The modeling framework is used to analyze the impacts of climate change-induced drought on water uses in the JRB. Given the uncertainty associated with future climate change, three alternate drought scenarios are developed to reflect a range of possible future water availability in the basin. Drought scenarios expressed as a percentage reduction of normal year water inflows are the following: mild (-22 percent), severe (-44 percent), and very severe (-66 percent). The characterization of drought scenarios severity is based on historical water inflows following the classification procedure of drought severity by the Jucar basin authority.

Estimations of climate change impacts in the Jucar basin indicate a reduction of water availability by 19 percent in the short-term (2010-2040), and 40 to 50 percent in the long-term (2070-2100) [*Ferrer et al.*, 2012]. A study by *CEDEX* [2010] forecasts water availability reductions between 5 and 12 percent for 2011-2040, between 13 and 18 percent for 2041-2070, and between 24 and 32 percent for 2071-2100. The drought scenarios considered in this paper cover the range of these estimations.

The model is used to assess the economic and environmental effects of alternative drought management policies under the drought scenarios described above. Three policy intervention alternatives are considered:

Institutional cooperation**:** Represents the current water management approach implemented in the JRB to cope with water scarcity and drought. This approach allows flexible adaptive changes in water allocations, based in the negotiation and cooperation of users. The special characteristic of this approach is that all water stakeholders are involved in the decision making process, and environmental concerns are considered.

Water market**:** There are increasing calls from international water institutions, water experts, and the Spanish government for market-based allocation of water during droughts. Water markets would allow water transfers between willing buyers and sellers, leading to welfare gains. This policy intervention highlights the question of whether these gains predicted by economic theory are quantitatively significant in practice. Under this policy, water trading is allowed among irrigation districts and with urban users in the JRB.

Water pricing**:** The water pricing approach is being implemented in the European water policies (European Commission 2012). Water pricing and water markets work well when water exhibits private good characteristics such as in urban networks, but not so well when water exhibits common pool resource or public good characteristics. There is a strong consensus among experts that water pricing could achieve sizable gains in efficiency and welfare in urban and industrial water networks (Hanemann 1997), although implementation could face technical and political difficulties. Irrigation water from surface watercourses and aquifers exhibits common pool resource characteristics, and the use of water pricing or other economic instruments requires transforming the resource into a private good. This transformation is quite difficult, especially in arid and semiarid regions under strong water scarcity pressures, and would require the support of stakeholders.

The model is well-suited to analyze other drought management policies, such as investments in irrigation technologies and increasing the allocation of water to the environment, but this policies are not analyzed here. The GAMS package has been used for model development and scenario simulation. The model has been solved using a mixed integer nonlinear programming algorithm.

6.2. Results

The economic and environmental outcomes from the three policy alternatives and drought scenarios are depicted in table 4. Further spatially disaggregated details for water use and benefits can be found in *Kahil et al.* [2014, Tables A3 and A4].

Institutional cooperation

The sum of private and environmental benefits in the JRB under the *Institutional cooperation* policy (baseline) and normal flow conditions amounts to 548 million € (Table 4). Water extractions are 1,119 Mm^3 divided between 1,000 Mm^3 in irrigation activities that generate 190 million €, and 119 Mm^3 in urban centers that generate 283 million € of economic surplus. About 60 Mm^3 of return flows from the ARJ and RB

Table 4. Alternative drought policies: institutional, water markets, and water pricing.

Drought Scenario	Normal Year	Mild Drought			Severe Drought		
Type of Policy	Institutional cooperation (Baseline)	Institutional cooperation	Water markets and Water pricing		Institutional cooperation	Water markets and Water pricing	
Water Use (Mm^3)							
Irrigation districts	1000	878	878		653	653	
Mancha Oriental	399	359	363		304	316	
Canal Jucar-Turia	125	102	120		77	116	
Acequia Real Jucar	200	180	197		131	185	
Escalona	33	30	32		18	31	
Ribera Baja	243	207	166		123	4	
Urban use	119	105	105		74	74	
Environmental flows (inflows to Albufera)	60	52	50		34	29	
Private and Environmental Benefits (million Euros)							
Type of Policy	Institutional cooperation (Baseline)	Institutional cooperation	Water markets	Water pricing	Institutional cooperation	Water markets	Water pricing
Private benefits							
Irrigation	190	171	175	93	136	148	54
Mancha Oriental	80	72	72	37	61	62	31
Canal Jucar-Turia	45	40	42	33	36	39	17
Acequia Real Jucar	34	31	32	17	23	25	4
Escalona	7	7	7	5	4	5	2
Ribera Baja	24	21	22	1	12	17	0
Urban use	283	276	276	276	241	241	241
Total	473	447	451	369	377	389	295
Environmental benefits	75	37	32	32	22	19	19
Private and environmental benefits	548	484	483	401	399	408	314

(Top) Water allocations to irrigation, urban use and environment in million cubic meters. (Bottom) Private benefits from irrigation and urban use, and environmental benefits in million Euros.

irrigation districts feed the Albufera wetland. These return flows support the good ecological status of the wetland. Environmental benefits provided by the Albufera wetland are 75 million €.

Results from drought scenarios indicate that drought events may reduce private and environmental benefits up to 140 million € (severe drought). Water use patterns show a reduction in extractions of surface water (up to 52%) and groundwater (up to 9%). The share of groundwater expands when drought increases in severity, from 42 percent in normal years up to 57 percent in very severe drought years. Irrigation activities face the main adjustment to water scarcity, with almost 90 percent of restrictions allocated to irrigation and the remainder allocated to urban uses.

The irrigation sector reduces surface water extractions up to 350 Mm^3 (severe drought). Increased pumping is allowed in the lower Jucar, while the curtailment of groundwater extractions is achieved in the EM irrigation district where farmers have been cooperating to control extractions during the last two decades. The reasons explaining this cooperation are the rising pumping costs from the very large aquifer depletion, and the significant pressures from downstream users losing water, and from the basin authority.

The losses of benefits to the irrigation sector under the *Institutional* policy range between 19 and 54 million € under mild and very severe drought conditions, and the irrigated area falls between 14,200 and 39,000 ha, respectively. By irrigation technology, the share of flood irrigation decreases while the share of sprinkler and drip irrigation increases. These changes in land use and irrigation technology distribution result in declining water application rates as drought severity intensifies.

Irrigation benefits in all five irrigation districts are reduced in drought years, but the impacts are distributed quite differently varying over space and severity of drought. Benefit losses in the traditional districts (ARJ, ESC, and RB) are larger than in the EM and CJT districts. Water use patterns show that the proportional cutback of surface water diversion during drought spells is lower in the traditional irrigation districts (ARJ, ESC, and RB), although with larger economic losses because they cannot totally substitute surface water with groundwater. The EM and CJT districts are based mostly on groundwater, which reduce their vulnerability to drought.

The cropping pattern and irrigation technology distribution by district and drought scenario can be found in *Kahil et al.* [2014, figures A2 and A3]. Results show the water and land management options for adapting to water scarcity, which are changes of crop mix, land fallowing, and improving irrigation efficiency through the adoption of water conserving-technologies. Generally, irrigation districts reduce the irrigated area of cereals and fruit trees, while maintaining the area of vegetables. By irrigation technology, the share of flood irrigation is reduced while the share of sprinkler and drip increases. However, the adaptive responses vary among the districts. Several factors may explain the varying adaptive responses of irrigation districts to increasing water scarcity. These are cropping patterns and crop diversification, the degree of irrigation modernization of the district, and the access to alternative water resources.

The reduction in irrigation water extractions has negative impacts on the Albufera wetland, which is mostly fed by irrigation return flows. Total irrigation return flows decrease up to 135 Mm^3, depending on the drought severity. Consequently, water inflows to the Albufera wetland dwindle – falling up to 26 Mm^3. Under severe drought conditions, water inflows to the Albufera wetland are less than the critical threshold E_l equal to 51 Mm^3, causing a regime shift in the ecosystem. Damages to the Albufera wetland under drought conditions are substantial and may exceed 70 percent of the benefit level in normal years.

The current water regulation in the JRB guarantees the priority of urban water for the human population. During severe drought spells the urban demand must be fully satisfied first because of such priority rules. The simulated drought scenarios show a reduced supply to the main cities in the JRB. However, the full demand of Valencia and Sagunto is always met with additional water from the bordering Turia River Basin. During extreme drought periods, the provision of water to these cities is supplied equally from the Jucar and the Turia Rivers. In the city of Albacete, the supply of water during dry periods is amended by pumping groundwater from the Eastern La Mancha aquifer [*CHJ*, 2009]. The simulation results for the urban sector indicate that the provision of surface water for urban use from the Jucar River falls by almost half, while groundwater extractions increase up to 8 Mm^3. The losses of benefits during droughts in the urban sector are nearly 15 percent in the worst-case scenario, because water provision is maintained with additional extractions from the Turia River and the Eastern La Mancha aquifer, but at higher costs. Several rationing measures have been implemented in the JRB to reduce water demand such as the installation of advanced water meters and the promotion of the use of water-saving devices [*CHJ*, 2009]. However, their effectiveness was quite limited, and they were not considered in our model.

Water markets

Results for the *Water market* policy indicate that introducing water trading in the JRB increases private benefits up to 3 percent compared to the *Institutional* policy (baseline). Irrigation benefits increase under water markets up to 9 percent, and urban benefits remain unchanged. This is explained because water trading occurs only among irrigation districts, and there is no water transfer to the urban sector. Irrigation water shadow prices in the market are greater than the cost of alternative water resources available to the urban sector in the JRB. Long run policy analysis may reorder these results because of possible changes in relative shadow prices of irrigation and urban water use.

Water trading becomes more pronounced as drought severity intensifies, with trades increasing from 1 Mm^3 (under a normal flow scenario) up to 119 Mm^3 (under very severe drought scenario). These results indicate that the benefits from implementing water markets are higher in drought situations compared to normal years.

In normal years, the gains from the *Water market* policy are modest compared to the *Institutional* policy, which means that the current institutional approach used in the JRB to allocate water among users is almost efficient. During drought periods, Pareto improvements could be achieved by allowing water trading among irrigation districts. Hence, introducing water markets in the JRB could mitigate drought damages for irrigation activities. Moreover, drought damages become more evenly distributed among irrigation districts in the *Water market* policy compared to the *Institutional* policy.

The water available under each drought scenario is the same for the *Institutional* and *Water market* policies. However, water markets increase consumption through crop evapotranspiration with additional reductions in return flows of up to 19 Mm^3 (10%) compared to the *Institutional* policy. These 19 Mm^3 of additional reductions are divided between 14 Mm^3 of return losses to the Jucar River and aquifers, and 5 Mm^3 of return losses to the Albufera wetland. Under the *Water market* policy, farmers maximize their benefits from water use by increasing crop evapotranspiration, either by increasing crop area, crop switching, or changing irrigation technology.

Under mild drought conditions, water inflows to the Albufera wetland are less than the critical threshold E_1 equal to 51 Mm^3, causing a shift in the ecosystem regime. The ecosystem regime shift takes place faster under the *Water market* policy compared to the *Institutional* policy. The reason is that the Albufera wetland is linked to the ARJ and RB irrigation districts that display a lower value of water than other districts. Under the drought scenarios, the ARJ and RB districts gain by selling water to other districts. As a consequence, return flows to the wetland under the *Water market* policy decline compared to the *Institutional policy*, leading to further desiccation and ecosystems degradation. Under severe and very severe droughts, the Albufera receives fewer inflows from the *Water market* policy than from the *Institutional* policy, but environmental benefits changes are small because they have already reached their lowest value. These results indicate that *Water market* reduces water availability to environmental uses, despite the fact that the small legally-required environmental flows are included in the hydro-economic model. However, the Albufera wetland does not have at present minimum binding inflows, and therefore receives less water under the *Water market* policy.

Water pricing

Water pricing in irrigation, to achieve water conservation, has been the subject of debate since the 1990s. A string of the literature finds that irrigation water pricing has limited effects on water conservation (Moore 1991, Sheierling et al. 2004), and some authors indicate that water markets seem far more effective than water pricing for allocating irrigation water (Cornish et al. 2004).

Water pricing is the worst policy option in terms of private and environmental benefits for the whole basin. Benefits of water pricing under drought are 20 percent lower than benefits of water markets or institutional policies (see table 4, last row). The water pricing instrument is politically unfeasible because farmers lose up to 60 percent of their profits when water pricing is used instead of the institutional or water markets alternatives, with profits falling to 93 and 54 million Euros under mild and severe drought, respectively (Table 4).

In summary, the main findings regarding the *Institutional*, *Water market* and *Water pricing* policies are the following. The private and environmental benefits under the current institutional cooperation are close to the benefits achieved under water markets

(Table 4). Under severe drought the difference in benefits is small between institutional cooperation and water markets (399 versus 408 million Euros; last row in table 4), and under mild drought the benefits are the same (484 million Euros). This demonstrates that the policy of water markets is not superior to the policy of institutional cooperation in Jucar. Furthermore, the policy of cooperation is more environmental friendly than pure water markets because the policy of cooperation allocates more water to the environment than water markets.

Another important finding is that the *Water pricing* policy is the worst possible alternative to address water scarcity and drought. Two reasons support this finding: first, the private and environmental benefits of water pricing are considerably below the benefits of institutional cooperation or water markets; and second, water pricing is politically unfeasible because choosing water pricing instead of the institutional or water market alternatives makes farmers loose 60 percent of their profits. These profits losses are the costs that farmers will sustain because of the wrong choice of policy. Since farmers are fully aware of their staggering costs of the water pricing alternative, they will strongly oppose this option leading to the failure of the water pricing policy.

7. Conclusions, policy implications, and future research

This paper presents the development and application of a policy-relevant integrated hydro-economic model. The contribution of this paper to previous hydro-economic modeling efforts stems from the development of a reduced form hydrological component, including theoretical concepts, data requirements, calibration, and use for climate and policy analysis. The idea is basically that when a detailed hydrological component is not available (which is the case in many basins worldwide), a calibrated reduced form can be used to predict water flows, becoming a component of hydro-economic modeling. Furthermore, the hydro-economic model includes a detailed regional economic component, and it accounts for ecosystem benefits in a way that makes them comparable with the benefits derived from other water uses. This modeling approach could be easily applied to most basins around the world.

The model has been used for empirical water policy analysis in a semiarid basin in Southeastern Spain, the Jucar River Basin, which is a good case for studying policies dealing with water scarcity and drought impacts from the impending climate change. The Jucar River is under severe stress, with acute water scarcity problems and escalating degradation of ecosystems. This is a common situation in many arid and semiarid basins around the world, and the empirical findings provide valuable insights to policy-makers not only in Spain but also in these arid and semiarid basins.

The implementation of a pure water market policy in the Jucar River Basin shows modest gains compared to the current institutional setting. Yet, the water market achieves a more even distribution of drought losses among irrigation districts. The reason could be that the current institutions involve asymmetric negotiation power among users in the basin authority. However, the water market entails a reduction of the water available to the environment, causing faster ecosystem regime shifts compared to what may happen under the current institutional setting. The reason is that water is mostly a common pool resource with environmental externalities, and markets disregard these externalities leading to excessive water extractions and damages to ecosystems.

Water pricing is the worst policy option to address water scarcity and droughts. This is so not only because it reduces the private and environmental benefits in the basin, but also because the cost of water pricing to farmers are staggering compared to the current institutional cooperation or to water markets. Therefore, it is impossible for farmers to accept the water pricing alternative, and the empirical findings demonstrate that this policy is doomed to failure.

The results highlight the advantages of negotiation and stakeholders' cooperation, which is the current institutional approach to water management in Spain. Indeed, compared to a pure water market policy (Pareto-efficient solution), this institutional approach achieves almost the same economic outcomes and better environmental outcomes. The policy implications of these findings highlight the importance of stakeholders' cooperation, and call for a reconsideration of water policies, specially water pricing. Water management arrangements and policies in arid and semiarid basins around the world are mostly based on command and control instruments or pure economic instruments, disregarding the potential of stakeholders' cooperation. These instruments fail because they lack legitimacy and knowledge of local conditions. The findings in the Jucar seem to indicate the importance of collective action in achieving a more sustainable water management.

A number of limitations of the hydro-economic model developed in this paper need to be addressed in future research. First, many questions linked to water and the environment involve time-dependent dynamic elements, such as groundwater depletion, reservoir storage, multi-year droughts, and river-aquifer interactions. The time dimension should be considered for an accurate assessment of climate change impacts and policy evaluation. Then, environmental benefits provided by the Albufera wetland to society are estimated, based on the quantity of water inflows from irrigation activities, disregarding other sources of water and quality variables. These limitations point to pathways by which future research could advance modeling performance to inform water policy at basin scale within the contexts of scarcity, droughts, and climate change. For now, this hydro-economic model illustrates how such a model can integrate the multiple dimensions of water resources, constituting a valuable tool to support the design of sustainable water management policies in arid and semiarid regions, as is the case of the Jucar River Basin.

Acknowledgements

This study has been financed by project BIL/13/MA/072 from MAPFRE and project INIA RTA2010-00109-C04 from the Spanish Ministry of Economy and Competitiveness. Special assistance has been received from Llorenç Avella, Marta García-Molla, and Manuel Pulido-Velazquez (UPV), Alfonso Calera and David Sanz (IDR), and Maria Calera (UCL).

References

Alcamo, J., T. Henrichs, and T. Rösch (2000), World water in 2025: global modeling and scenario analysis for the World Commission on Water for the 21st Century, Kassel World Water Series, Report 2, Center for Environmental System Research, University of Kassel, Kassel.

Booker, J. and R. Young (1994), Modeling intrastate and interstate markets for Colorado River water-resources. *J. Environ. Econ. Manag.,* 26 (1): 66–87.

Booker, J., A. Michelsen, and F. Ward (2005), Economic impact of alternative policy responses to prolonged and severe drought in the Rio Grande Basin. *Water Resour. Res.*, 41 (2): 1-15. doi:10.1029/2004WR003486

Booker, J., R. Howitt, A. Michelsen, and R. Young (2012), Economics and the modeling of water resources and policies. *Nat. Resour. Model.*, 25(1): 168-218.

Brander, L., R. Florax, and J. Vermaat (2006), The empirics of wetland valuation: a comprehensive summary and a meta-analysis of the literature. *Environ. Resour. Econ.*, 33: 223-250.

Cai, X., D. McKinney, and L. Lasdon (2003), Integrated Hydrologic-Agronomic-Economic Model for River Basin Management. *J. Water Res. Pl. Manag.*, 129: 4-17.

Centro de Estudios y Experimentación de Obras Públicas (CEDEX) (2010), Estudio de los impactos del cambio climático en los recursos hídricos y las masas de agua, Ficha

1: Evaluación del impacto del cambio climático en los recursos hídricos en régimen natural, MARM, Madrid.

Collazos, G. (2004), Sistema soporte de decisión para evaluación y optimización económica de sistemas de recursos hídricos, Ph.D. Thesis, Universidad Politécnica de Valencia, Valencia.

Confederación Hidrográfica del Júcar (CHJ) (2002), Estudio de utilización conjunta de los recursos hídricos superficiales y subterráneos en las cuencas media y baja de los ríos Júcar y Turia, Informe final, Documento de síntesis, MIMAM, Valencia.

Confederación Hidrográfica del Júcar (CHJ) (2007), Plan especial de Alerta y eventual sequia en la Confederacion Hidrografica del Júcar, MIMAM, Valencia.

Confederación Hidrográfica del Júcar (CHJ) (2009), Esquema provisional de temas importantes, MARM, Valencia.

Confederación Hidrográfica del Júcar (CHJ) (2012), Sistema Júcar: Datos del año hidrológico 2012-2013, MAGRAMA, Valencia.

Cornish, G., B. Bosworth, C. Perry and J. Burke (2004), Water charging in irrigated agriculture. An analysis of international experience. FAO Water Reports 28. FAO, Rome.

Deidda, D. (2004), Análisis económico de sistemas de recursos hídricos mediante modelos de simulación: Aplicación a la cuenca del rio Júcar, M.S. Thesis, Universidad Politécnica de Valencia, Valencia.

Del Saz, S. and L. Perez (1999), El valor de uso recreativo del parque natural de L'Albufera a través del método indirecto del coste de viaje. *Estud. Econ. Apli.*, 11: 41-62.

Esteban, E. and J. Albiac (2012), The problem of sustainable groundwater management: the case of La Mancha aquifers, Spain. *Hydrogeol. J.,* 20(5): 851-863.

European Commission (EC) (2007), Communication from the Commission to the European Parliament and the Council, Addressing the challenge of water scarcity and droughts in the European Union, COM 414/2007, European Commission, Brussels.

European Commission (EC) (2012), Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. A Blueprint to Safeguard Europe's Water Resources. COM(2012) 673 final. Brussels: European Commission.

Federal Emergency Management Agency (FEMA) (1995), National mitigation strategy: partnerships for building safer communities, Federal Emergency Management Agency, Washington DC.

Ferrer, J., M. Perez, S. Jiminez, T. Estrela, and J. Andreu (2012), GIS-based models for water quantity and quality assessment in the Júcar River Basin, Spain, including climate change effects. *Sci. Total Environ.*, 440: 42-59.

García-Molla, M., C. Sanchis, M. Ortega, and L. Avella (2013), Irrigation associations coping with drought: The case of four irrigation districts in Eastern Spain, In Drought in Arid and Semi-Arid Environments: A Multi-Disciplinary and Cross-Country Perspective, edited by K. Schwabe, J. Albiac, J. Connor, R. Hassan and L. Meza, pp. 101-122, Springer, Dordrecht.

Generalitat Valenciana (GV) (2009), Base de datos 1T de superficies de cultivos por término municipal para La Generalitat Valenciana 2009, Departamento de Agricultura, Generalitat Valenciana. Valencia.

Gobierno de Castilla La Mancha (CCLM) (2009), Base de datos 1T de superficies de cultivos por término municipal para Castilla La Mancha 2009, Departamento de Agricultura, Gobierno de Castilla La Mancha, Albacete.

Harou, J., M. Pulido, D.E. Rosenberg, J. Medellín, J. Lund, and R. Howitt (2009), Hydro-economic models: Concepts, design, applications, and future prospects. *J. Hydrol.*, 375: 627–643.

Intergovernmental Panel on Climate Change (IPCC) (2007), Climate change 2007: Synthesis report, Contribution of Working Groups I, II and III to the Fourth Assessment Report of the IPCC, IPCC, Geneva.

Instituto Geológico y Minero de España (IGME) (2009), Análisis de la evolución y del estado de las masas de agua subterránea utilizadas en épocas de sequia en la cuenca del Júcar, Informe de situación final de la campaña de riego, IGME, Madrid.

Instituto Nacional de Estadística (INE) (2009), Censo agrario 2009, INE, Madrid.

Jorgensen, S., F. Xu, J. Marques, and F. Salas (2010), Application of indicators for the assessment of ecosystem health, In Handbook of Ecological Indicators for Assessment of Ecosystem Health, edited by S. Jorgensen, F. Xu, and R. Costanza, pp 9-75, CRC press, Boca Raton.

Kahil, M.T., A. Dinar, and J. Albiac (2014), Modeling water scarcity and drought severity for policy adaptation to climate change: Application to the Jucar Basin, Spain, University of California Water Science and Policy Center, Working Paper 01-0114, January 2014. http://wspc.ucr.edu/working_papers/WSPC_WP_01_0114_policy%20climate%20change%20jucar%20basin.pdf

Keeler, B., S. Polasky, K. Brauman, K. Johnson, J. Finlay, A. O'Neill, K. Kovacs, and B. Dalzell (2012), Linking water quality and well-being for improved assessment and valuation of ecosystem services. *PNAS*, 109(45): 18619-18624.

Kirby, M., R. Bark, J. Connor, E. Qureshi, and S. Keyworth (2014), Sustainable irrigation: How did irrigated agriculture in Australia's Murray–Darling Basin adapt in the Millennium Drought?. *Agr. Water Manage*, in Press.

Krause, P., D. Boyle, and F. Bäse (2005), Comparison of different efficiency criteria for hydrological model assessment. *Advances in Geosciences*, 5: 89–97.

Letcher, R., B. Croke, and A. Jakeman (2007), Integrated assessment modelling for water resource allocation and management: A generalized conceptual framework. *Environ. Modell. Softw.*, 22: 733-742.

Lund, J., X. Cai, and G. Characklis (2006), Economic engineering of environmental and water resource systems. *J. Water Res. Pl.*, 132 (6): 399–402.

Maneta, M., M. Torres, W. Wallender, S. Vosti, R. Howitt, L. Rodrigues, L. Bassoi, and S. Panday (2009), A spatially distributed hydroeconomic model to assess the effects of drought on land use, farm profits, and agricultural employment. *Water Resour. Res.,* 45:1-19. doi:10.1029/2008WR007534

McKinney, D., X. Cai, M. Rosegrant, C. Ringler, and C. Scott (1999), Modeling water resources management at the basin level: review and future directions, SWIM Paper 6, International Water Management Institute, Colombo.

Ministerio de Medio Ambiente, Rural y Marino (MARM) (2010), Análisis de la economía de los sistemas de producción: Resultados técnico-económicos de explotaciones agrícolas de en Valencia y Castilla La Mancha en 2009, Subsecretaria de Medio Ambiente, Rural y Marino, MARM, Madrid.

Moore, M. (1991), The bureau of reclamations new mandate for irrigation water conservation – purposes and policy alternatives. *Water Resour. Res.*, 27(2): 145–155.

National Oceanic and Atmospheric Administration (NOAA) (2008), Summary of National Hazard Statistics for 2008 in the United States, National Weather Service, NOAA, Washington DC.

Önal, H. and B. McCarl (1991), Exact aggregation in mathematical programming sector models. *Can. J. Agr. Econ.,* 39: 319-334.

Pulido, M., J. Andreu, A. Sahuquillo, and D. Pulido-Velazquez (2008), Hydro-economic river basin modelling: The application of a holistic surface–groundwater model to assess opportunity costs of water use in Spain. *Ecol. Econ.*, 66: 51-65.

Sanchis, C. (2011), La Albufera de Valencia: cincuenta años de eutrofia. *Métode* 70.

Sanz, D., S. Castaño, E. Cassiraga, A. Sahuquillo, J. Gómez, S. Peña, and A. Calera (2011), Modeling aquifer-river interactions under the influence of groundwater abstractions in the Mancha Oriental System (SE Spain). *Hydrogeol. J.*, 19: 475-487.

Scheffer, M., S. Carpenter, J. Foley, and B. Walker (2001), Catastrophic shifts in ecosystems. *Nature,* 413:591-596.

Scheierling, S., R. Young and G. Cardon (2004), Determining the price responsiveness of demands for irrigation water deliveries versus consumptive use. *Journal of Agricultural and Resource Economics* 29(2): 328–345.

Turner, K., J. van den Bergh, T. Söderqvist, A. Barendregt, J. van der Straaten, E. Maltby, and E. van Ierland (2000), Ecological-economic analysis of wetlands: scientific integration for management and policy. *Ecol. Econ.*, 35:7-23.

Vörösmarty C., P. McIntyre, M. Gessner, D. Dudgeon, A. Prusevich, P. Green, S. Glidden, S. Bunn, C. Sullivan, C. Liermann, and P. Davies (2010), Global threats to human water security and river biodiversity. *Nature*, 467: 555-561.

Ward, F. (2009), Economics in integrated water management. *Environ. Modell. Softw.*, 24: 948-958.

Woodward, R. and Y. Wui (2001), The economic value of wetland services: A meta-analysis. *Ecol. Econ.*, 37: 257–270.

Young, R. and J. Loomis (2014), Determining the economic value of water: Concepts and methods. 2nd edition, RFF Press and Routledge, Oxon.