Modeling water scarcity and droughts for policy adaptation to climate change in arid and semiarid regions

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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. 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 to overcome the negative economic effects of droughts, although the environmental effects may weaken its advantages for society. The environmental water market policy, where water is acquired for the environment, is an appealing policy to reap the private benefits of markets while protecting ecosystems. 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 arid and semiarid 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.

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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 sixfold 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% of the world population living in regions with severe water scarcity. Furthermore, about 65% 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% 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% of the value of irrigated agriculture, representing about 1% 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 (1994), McKinney et al. (1999), Cai et al. (2003), Booker et al. (2005), Pulido et al. (2008), Molinos et al. (2014), and Ward (2014). 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. 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.

The integrated model simulates demand nodes' behavior under different drought scenarios (mild, severe, and very severe drought) and policy intervention alternatives (baseline or 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, land use decisions, and private and social benefits and costs of water utilization. The hydro-economic model is empirically tested in an arid and semiarid basin in Southeastern Spain, the Jucar River Basin. 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 (Cornish et al., 2004; Varela et al., 2011; Connor and Kaczan, 2013).

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 toward 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 Fig. 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, 2014).
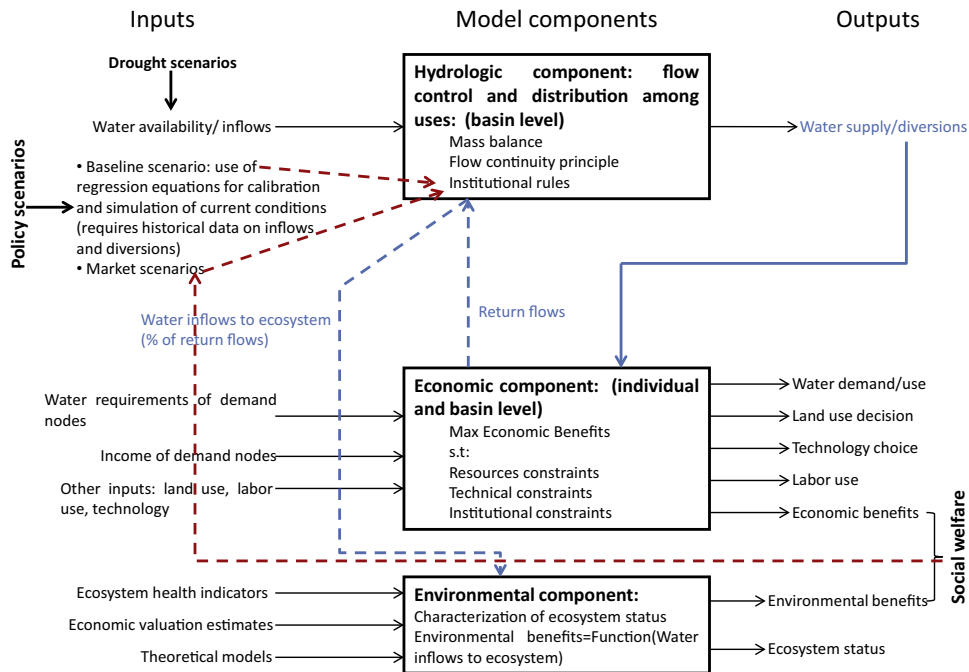
Fig. 1. Modeling framework.
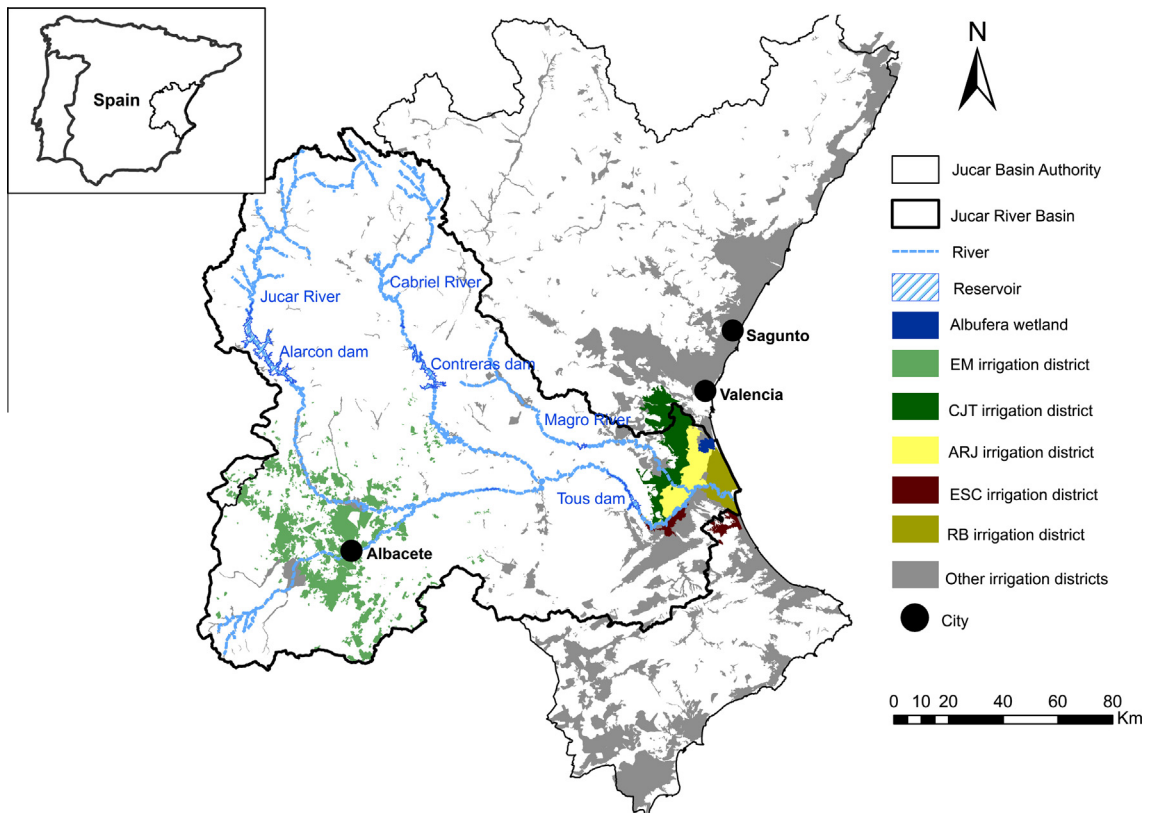
Fig. 2. Map of the Jucar River Basin.

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

Fig. 3. JRB network.

infrastructure features at each node, including facility capacities, losses, and evaporation.

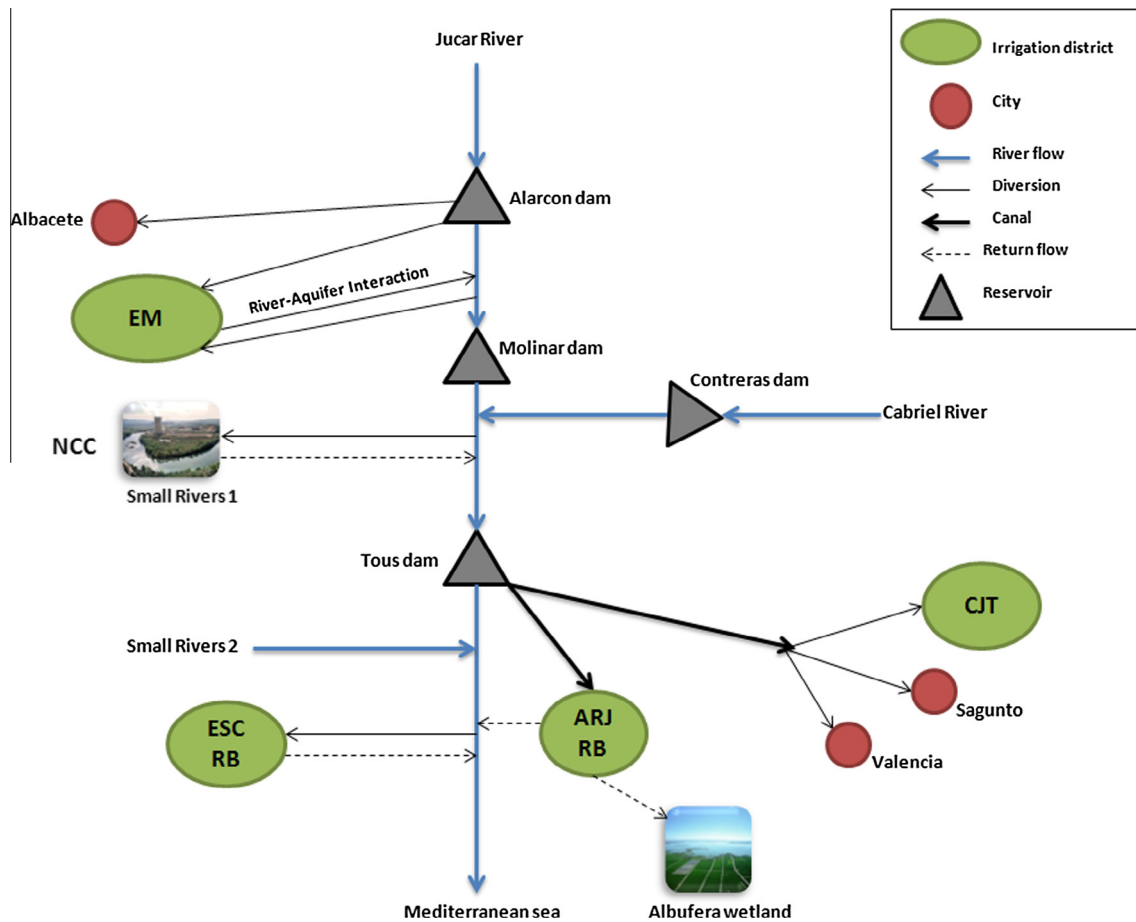The reduced form hydrological model allows controlling the flows of water in each node and estimating the distribution of the available water among users under each climate condition. The model is calibrated so that predicted allocations to users in both normal and drought periods match historical water 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 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 Fig. 1 and it is applied to the drought management problem in an arid and 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

Recently, signs have been mounting on successful water management approach in Spain (Schwabe et al., 2013). This approach is institutional and relies on the river basin authorities. There is a strong tradition of cooperation among water stakeholders within basin authorities in Spain dating back centuries. 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 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 (Fig. 2). It extends over 22,300 km² 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 1700 M m³, of which 930 are surface water and 770 are groundwater resources. Water extractions are 1680 M m³, very close to renewable resources, making the JRB an almost closed water system. Extractions for irrigated agriculture are nearly 1400 M m³. Urban and industrial extractions total 270 M m³, 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 2500 M m³. The aquifer is linked to the Jucar River stream, and it fed the Jucar River with about 150 M m³/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 M m³/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 m³/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 M m³ 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 (García-Molla et al., 2013).

The Albufera wetland is a freshwater lagoon with an area covering 2430 ha, supporting very rich aquatic ecosystems. Since 1989, the Albufera was catalogued in the RAMSAR list, and was declared a special protected area for birds. The Albufera receives water from the return flows of the irrigation districts in the lower Jucar, mainly from the ARJ and the RB districts. Other flows originate from the Turia River Basin, and from the discharge of untreated and treated urban and industrial wastewaters in the adjacent municipalities. At present, the Albufera 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 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).

4. Model components and scenarios

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 on the model and data sources).

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. Fig. 3 presents the hydrological network of the basin, including the most important infrastructures, and water supply and demand nodes.

The reduced form hydrological model estimates 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:

$$\text{Wout}_d = \text{Win}_d - \text{Wloss}_d - \text{Div}_d^{\text{IR}} - \text{Div}_d^{\text{URB}} \tag{1}$$

$$\text{Win}_{d+1} = \text{Wout}_d + r_d^{\text{IR}} \cdot \left(\text{Div}_d^{\text{IR}}\right) + r_d^{\text{URB}} \cdot \left(\text{Div}_d^{\text{URB}}\right) + \text{RO}_{d+1} \tag{2}$$

$$\text{Wout}_d \geqslant E_d^{\min} \tag{3}$$

The mass balance Eq. (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 $\text{Div}_d^{\text{URB}}$. The continuity Eq. (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 $\left[r_d^{\text{IR}} \cdot \left(\text{Div}_d^{\text{IR}}\right)\right]$, the return flows from the cities $\left[r_d^{\text{URB}} \cdot \left(\text{Div}_d^{\text{URB}}\right)\right]$, and runoff entering that river reach from tributaries, RO_{d+1}. Eq. (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 $\text{Div}_d^{\text{URB}}$, 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 $\left(\text{Div}_d^{\text{IR}}\right.$ or $\text{Div}_d^{\text{URB}}$, as dependent variables$\left.\right)$ 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 last 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 policy) 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 policy 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 coefficients from 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 1984–2004. 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:

$$\text{Max}B_k^{\text{IR}} = \sum_{ij} C_{ijk}' \cdot X_{ijk} \tag{4}$$

subject to

$$\sum_i X_{ijk} \leqslant \text{Tland}_{kj}; \quad j = \text{flood, sprinkler, drip} \tag{5}$$

$$\sum_{ij} W_{ijk} \cdot X_{ijk} \leqslant \text{Twater}_k \tag{6}$$

$$\sum_{ij} L_{ijk} \cdot X_{ijk} \leqslant \text{Tlabor}_k \tag{7}$$

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

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

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. Cereals can be irrigated using flood and sprinkler systems, and vegetables and fruit trees can be irrigated using flood and drip systems.

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, 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:

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	€/m³
Average urban water price	0.71	€/m³
Inverse water demand functions for cities		
Intercept (a_{du})		
Valencia	6	€
Albacete	6	€
Sagunto	6	€
Slope (b_{du})		
Valencia	−0.06	€/M m³
Albacete	−0.3	€/M m³
Sagunto	−0.5	€/M m³
Benefit function of the Albufera from water inflows		
Intercept (δ_1)	33	10⁶ €
First threshold of inflows to the Albufera (E_1)	51	M m³
Intercept (δ_2)	−214	10⁶ €
Slope (ρ_2)	4.8	€/m³
Second threshold of inflows to the Albufera (E_2)	78	M m³
Intercept (δ_3)	43	10⁶ €
Slope (ρ_3)	1.8	€/m³
Third threshold of inflows to the Albufera (E_3)	138	M m³
Economic value of the Albufera wetland	13,600	€/ha

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

subject to

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

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

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. Eq. (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 paper 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.

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:

$$E_{\text{Albufera}} = \alpha \cdot r_{\text{ARJ}}^{\text{IR}} \cdot \left(\text{Div}_{\text{ARJ}}^{\text{IR}} \right) + \beta \cdot r_{\text{RB}}^{\text{IR}} \cdot \left(\text{Div}_{\text{RB}}^{\text{IR}} \right) \quad (13)$$

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

where Eq. (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 $\left[r_{\text{ARJ}}^{\text{IR}} \cdot \left(\text{Div}_{\text{ARJ}}^{\text{IR}} \right) \right]$ and $\left[r_{\text{RB}}^{\text{IR}} \cdot \left(\text{Div}_{\text{RB}}^{\text{IR}} \right) \right]$ are return flows from the ARJ and RB irrigation districts, respectively. Eq. (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.

The empirical benefit function of the Albufera wetland has been developed in two steps. First, time series data of various ecosystem health indicators of the wetland have been collected, including the quantity of water inflows, the number of water replenishments, *chlorophyll a* and phosphorus concentrations, and salinity levels. These indicators are used to calculate a unique health index of the wetland for each year of available data, following the methodology developed by Jorgensen et al. (2010). The health index ranges between 0 (bad ecological status) and 1 (good ecological status). Once the health index for each year is calculated, then thresholds E_1 and E_2 under which the ecosystem status changes significantly are determined.

Second, the information on the economic value of the wetland is only available for one year. The value of this particular year is extrapolated to the other years as a linear function of the health index of each year. This linear extrapolation assumes that the environmental benefits of the wetland are a function of its ecosystem health. Once the economic values are calculated for each year, 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 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 of the Albufera and the parameter estimates of the benefit function 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:

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

subject to the constraints in Eqs. (1)–(3), (5)–(9), and (11)–(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 4.5:

$$\text{Div}_d^l = f(\text{Win}_d) \; \forall \; l, d \quad (16)$$

$$\sum_{ld} \text{Div}_d^l \leqslant \overline{W} \quad (17)$$

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. Eqs. (16) and (17) are used to allocate water among users under the baseline policy (institutional approach). Eq. (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. This equation incorporates the institutional intervention in water allocations. Eq. (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, the allocations to users are determined fully by maximizing the entire basin's benefits (Eq. (15)), subject to the total basin water availability (Eq. (17)). The regression equations (Eq. (16)) are removed from the model. Therefore, water is allocated to the higher-value uses (efficient allocation) without any institutional intervention in allocations. The labor constraint (7) is relaxed to allow labor transfers among irrigation districts. The market price of water is determined endogenously in the model based on the shadow value of water.

4.5. 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%), severe (−44%), and very severe (−66%). 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% in the short-term (2010–2040), and 40–50% in the long-term (2070–2100) (Ferrer et al., 2012). A study by CEDEX (2010) forecasts water availability reductions between 5% and 12% for 2011–2040, between 13% and 18% for 2041–2070, and between 24% and 32% 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.

Baseline policy: 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 between users. The special characteristic of this approach is that all water stakeholders are involved in the decision making process, and environmental concerns are considered.

Ag-Urban 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.

Environmental water market: In recent decades, the water market policy to acquire water for the environment has been gaining ground in some parts of the world, such as in Australia and the United States. This policy consists of having the basin authority participating in the water market to acquire water for the Albufera wetland. As such, the wetland is competing for water with other users and does not depend passively on remaining return flows.

The reason for having two separate policies for water trading (Ag-Urban, and Environment) is mainly because of the nature of agents involved. While in the Ag-Urban water market the traders are private decision makers, the water for environmental purposes has the public agency as a steward for the environment, which sometimes creates conflicts with the other sectors. The GAMS package has been used for model development and scenario simulation. The model has been solved using a mixed integer nonlinear programming algorithm.

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 (Andreu et al., 1996; CHJ, 2002, 2012; Collazos, 2004) (Fig. 4).

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 (Fig. 4).

Return flows have been calculated as the fraction of diverted water not used in crop evapotranspiration $\left[r_d^{IR} \cdot \left(\text{Div}_d^{IR} \right) \right]$ and urban consumption $\left[r_d^{URB} \cdot \left(\text{Div}_d^{URB} \right) \right]$. Most return flows originate from irrigation, with overall irrigation efficiency estimated at 60%, 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 Albufera receives 28 and 23% 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 policy for the hypothetical future drought scenarios.

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, indicating that the model reproduces adequately the hydrologic conditions (Table 3). A detailed description of the validation process can be found in Kahil et al. (2014).

6. Results and discussion

The economic and environmental outcomes from the three policy alternatives and drought scenarios are depicted in Tables 4–6. Further spatially disaggregated details for water use and benefits can be found in Kahil et al. (2014, Tables A3 and A4).

6.1. Baseline policy

Social welfare, which is the sum of private and environmental benefits, in the JRB under the *Baseline policy* and normal flow conditions amounts to 548 million € (Table 4). Water use is 1149 M m³, of which 672 is surface water and 477 is groundwater resources (Table 5). Irrigation activities generate 190 million € from using 1030 M m³. The social surplus of urban centers is 283 million € and they use 119 M m³. About 60 M m³ of return flows from the ARJ and RB irrigation districts feed the Albufera wetland, which 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 social welfare in the JRB up to 138 million €. 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% in normal years up to 57% in very severe drought years. Irrigation activities face the main adjustment to water scarcity, with almost 90% of restrictions allocated to irrigation and the remainder allocated to urban uses.

The irrigation sector reduces surface water extractions up to 296 M m³ and groundwater extractions up to 52 M m³. 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 benefit losses to the irrigation sector in the *Baseline policy* range between 19 and 55 million € under mild and very severe drought conditions, and the irrigated area is reduced by 14,200 and 39,000 ha, respectively. 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 decreases while the share of sprinkler and drip irrigation increases (Table 6). These changes in land use and irrigation technology distribution result in declining water application rates as drought severity intensifies.
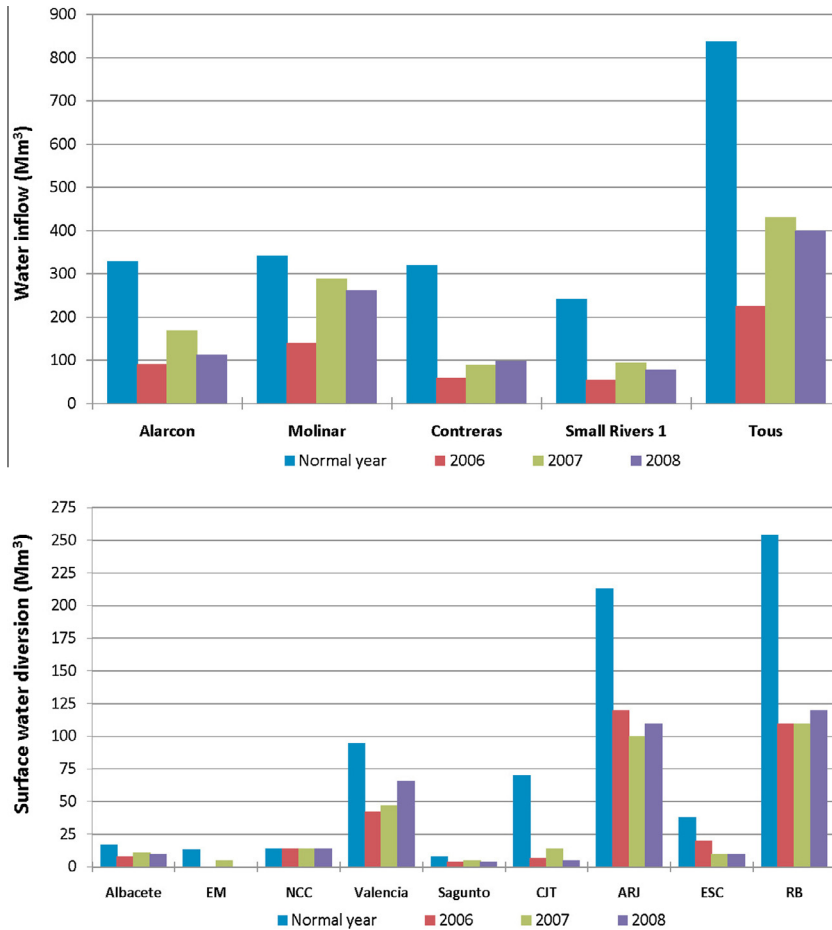
Fig. 4. Surface water inflows to the main reservoirs and river reaches (top) and diversions for the demand nodes (down) in the Jucar River Basin.

Table 2
Relationships between water diversions and inflows.

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

Note: $Win_{Alarcon}$ = water inflows to Alarcon dam; Win_{Tous} = water inflows to Tous dam; Win_{SR1} = water inflows from small rivers 1; $r^{IR} \cdot Div^{IR}$ = 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 $p < 0.2$.

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. However,

Table 3
Comparison between simulated (Sim) and observed (Ob) water diversions (M m^3).

Demand nodes	Normal flow year		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

Table 4
Benefits and irrigation labor use under the policy and drought scenarios.

Aggregate results	Normal flow	Mild drought	Severe drought	Very severe drought
Baseline policy				
Private benefits (10^6 €)				
Irrigation sector	190.3	170.9	152.7	135.4
Urban sector	282.6	276.3	266.4	240.9
Total	472.9	447.2	419.1	376.3
Environmental benefits (10^6 €)	74.7	37.2	33.0	33.0
Social welfare (10^6 €)	547.6	484.4	452.1	409.3
Irrigation labor use (Jobs)[*]	15,100	13,815	12,500	11,230
Ag-Urban water market				
Private benefits (10^6 €)				
Irrigation sector	190.5	174.9	161.2	147.5
Urban sector	282.6	276.3	266.4	240.9
Total	473.1	451.2	427.6	388.4
Environmental benefits (10^6 €)	74.7	33.0	33.0	33.0
Social welfare (10^6 €)	547.8	484.2	460.6	421.4
Irrigation labor use (Jobs)	15,110	14,350	13,620	12,830
Environmental water market				
Private benefits (10^6 €)				
Irrigation sector	195.4	180.2	165.2	160.1
Urban sector	282.6	276.3	266.4	240.9
Total	478.0	456.5	431.6	401.0
Environmental benefits (10^6 €)	277.6	275.9	272.6	255.7
Social welfare (10^6 €)	755.6	732.4	704.2	656.7
Irrigation labor use (Jobs)	14,610	13,720	12,440	10,560

[*] 1 job unit = 1920 h/year.

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 M m^3, depending on the drought severity. Consequently, water inflows to the Albufera wetland dwindle – falling up to 26 M m^3. Under severe drought conditions, water inflows to the Albufera wetland are less than the critical threshold E_1 equal to 51 M m^3, causing a regime shift in the ecosystem. Damages to the Albufera wetland under drought conditions are substantial and may exceed 50% of normal years benefit level.

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 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 M m^3. The losses of benefits during droughts in the urban sector are nearly 15% 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 were also 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.

6.2. Ag-Urban water market

Results for the *Ag-Urban water market* policy indicate that introducing water trading in the JRB increases private benefits up to 3% compared to the *Baseline policy*. Irrigation benefits increase under

Table 5
Water use and return flows under the policy and drought scenarios (M m^3).

Aggregate results	Normal flow	Mild drought	Severe drought	Very severe drought
Baseline policy				
Water use				
Irrigation sector	1030	908	793	683
Urban sector [*]	119	105	90	74
Total	1149	1013	883	757
Irrigation return flows				
Return flows to river and aquifers	267	231	195	158
Return flows to Albufera	60	52	43	34
Total	327	283	238	192
Ag-Urban water market				
Water use				
Irrigation sector	1030	908	793	683
Urban sector	119	105	90	74
Total	1149	1013	883	757
Traded water	1	41	87	119
Irrigation return flows				
Return flows to river and aquifers	267	224	183	144
Return flows to Albufera	60	50	40	29
Total	327	274	223	173
Environmental water market				
Water use				
Irrigation sector	936	801	672	546
Urban sector	119	105	90	74
Total	1055	906	762	620
Traded water	95	148	169	201
Irrigation return flows				
Return flows to river and aquifers	232	184	135	88
Return flows to Albufera	49	38	23	7
Total	281	222	158	95
Inflows to Albufera from trade	89	100	115	131

[*] The quantity of urban water use shown in the table represents only the part of supply from the JRB. During droughts, the urban sector uses additional quantity of water from the Turia River to cover the demand of Valencia and Sagunto.

Table 6
Land use under the policy and drought scenarios.

Aggregate results	Normal flow	Mild drought	Severe drought	Very severe drought
Baseline policy				
Irrigated area (ha)	156,830	142,615	130,530	117,780
Cropping pattern (ha)				
Cereals	70,430	63,460	58,060	52,055
Vegetables	22,540	20,090	18,390	16,720
Fruit trees	63,860	59,065	54,080	49,005
Irrigation system share (%)				
Flood	18	17	15	14
Sprinkler	37	37	38	38
Drip	45	46	47	48
Ag-Urban water market				
Irrigated area (ha)	156,900	144,520	134,490	124,040
Cropping pattern (ha)				
Cereals	70,420	62,760	56,590	50,400
Vegetables	22,550	20,340	18,890	17,430
Fruit trees	63,930	61,420	59,010	56,210
Irrigation system share (%)				
Flood	18	16	14	12
Sprinkler	37	37	38	38
Drip	45	47	48	50
Environmental water market				
Irrigated area (ha)	151,680	138,460	126,380	112,380
Cropping pattern (ha)				
Cereals	66,910	58,850	53,030	48,130
Vegetables	22,210	20,060	18,470	16,730
Fruit trees	52,560	59,550	54,880	47,520
Irrigation system share (%)				
Flood	17	14	11	8
Sprinkler	38	39	40	42
Drip	45	47	49	50

water markets up to 9%, and urban benefits remain unchanged. The reason is that 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 M m^3 (under a normal flow scenario) up to 119 M m^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 *Ag-Urban water market* policy are modest compared to the *Baseline 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 *Ag-Urban water market* policy compared to the *Baseline policy*.

The water available under each drought scenario is the same for the *Baseline* and *Ag-Urban water market* policies. However, water markets increase consumption through crop evapotranspiration with additional reductions in return flows of up to 19 M m^3 (10%) compared to the *Baseline* policy. These 19 M m^3 of additional reductions are divided between 14 M m^3 of return losses to the Jucar River and aquifers, and 5 M m^3 of return losses to the Albufera wetland. Under the *Ag-Urban 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 M m^3, causing a shift in the ecosystem regime. The ecosystem regime shift takes place faster under the *Ag-Urban water market* policy compared to the *Baseline 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 *Ag-Urban water market* policy decline compared to the *Baseline policy*, leading to further desiccation and ecosystems degradation.

Social welfare in the JRB under mild drought conditions decrease with the *Ag-Urban water market* policy compared to the *Baseline policy*. Under severe and very severe droughts, the Albufera receives fewer inflows from the *Ag-Urban water market* policy than from the *Baseline policy*, but environmental benefits remain unchanged because they have already reached their lowest value. These results indicate that *Ag-Urban 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 *Ag-Urban water market* policy.

6.3. Environmental water market

Under the *Environmental water market* policy, the basin authority operates in the water markets to purchase water for the Albufera wetland in order to maximize social welfare. Results indicate that basin's irrigation benefits may increase (up to 18%) compared to the *Baseline policy*. By introducing the *Environmental water market* policy, drought damages become more evenly distributed

among irrigation districts, and the traditional irrigation districts (ARJ, ESC, and RB) become much less vulnerable to droughts compared to the *Baseline policy*.

Irrigation water use decreases up to 20% compared to the *Baseline policy*. Irrigation water is more efficiently used under the *Environmental water market* policy compared to the *Baseline* and *Ag-Urban water market* policies. However, return flows fall significantly up to 51% reducing the Jucar River streamflows, aquifer recharge and return flows to the Albufera. The traded volume of water increases as drought severity intensifies from 95 M m^3 under normal flow scenario to 201 M m^3 under very severe drought. Further, the traded volume of water increases in the *Environmental water market* policy compared to the *Ag-Urban water market* policy to meet growing environmental and irrigation demand.

Water allocated to the Albufera wetland coming from irrigation in the market is between 89 and 131 M m^3, securing always a fixed amount of water (138 M m^3) flowing to the wetland. This amount is well above the minimum environmental requirements of the Albufera wetland set by the basin authority (60 M m^3), and thus ensures its good ecological status. Environmental benefits provided by the Albufera wetland to society increase considerably, and so does the social welfare of the JRB. Water reallocated from crops with low to high marginal value of water is between 6 and 70 M m^3.

Under the *Environmental water market* policy, the irrigated area falls in all drought scenarios (up to 5%) compared to the *Baseline policy*. The areas of cereals and fruit trees are reduced, while the area of vegetables remains broadly unchanged. For irrigation technology, the share of flood irrigation falls significantly, while the share of sprinkler and drip irrigation increases. As a consequence of the fall of land under production, irrigation labor use declines compared to the *Baseline policy*.

The results of the *Environmental water market* policy depend on the economic valuation of the Albufera wetland assumed in the empirical application. A sensitivity analysis has been conducted in order to assess the results from the *Environmental water market* policy, and their robustness to different economic valuation estimates of the wetland (see Table 11 in Kahil et al., 2014). Results do not change until the economic valuation estimate is changed by a factor of 25, from 13,600 €/ha estimate to 340,000 €/ha (high) and 544 €/ha (low).

The Albufera wetland already receives the optimal inflow (the maximum allowed in the model) for the 13,600 €/ha estimate, and for higher valuation estimates there is no need to purchase more water from the irrigation districts. This implies that the baseline ecosystem value is high enough to convince society to prioritize ecosystem health rather than damaging it. However, a lower ecosystem value modifies the outcome from the *Environmental water market* policy. Water inflows to the Albufera wetland fall for the low valuation estimate, and less water is purchased from the irrigation districts upsetting consequently the farmers' private benefits from selling water. These results call for an accurate valuation of the ecosystem services provided to society by the wetland, in order to avoid misleading decisions with respect to ecosystem protection.

7. Conclusions and policy implications

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, 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 an arid and 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 show 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.

Having the basin authority operating in the water market to acquire water for the Albufera wetland seems to be an appealing policy to keep up with the basin's increasing demand for water and to correct the pure market failure. The main effects of such a policy are improved social and private benefits of the basin, reduced vulnerability of irrigation districts to droughts, and a secure, fixed amount of water flowing to the Albufera wetland that ensures its good ecological status. Some negative effects include substantial decreases of the Jucar River streamflows and aquifer recharge, and the fall of employment in irrigation.

The empirical 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. 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 River Basin seem to indicate the importance of collective action in achieving a more sustainable water management. But these results do not imply that one type of policy instrument is superior to others for advancing sustainable water management under all circumstances. Some authors warn against the use of a single type of policy instrument (panacea) for solving water management problems (Ostrom et al., 2007). Water markets and collective action are alternative approaches to achieve welfare gains in the form of private and social benefits. Both approaches are intertwined though, because the water trading experiences worldwide indicate that pure markets tend to disregard third party effects, including environmental impacts. Well functioning water markets would require a great deal of regulation or cooperation by stakeholders within a strong institutional setting. Conversely, the institutional approach in countries such as Spain would work better by using carefully-designed economic instruments. These incentives would introduce more flexibility into the institutional process of decision making and implementation.

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