TESIS DE LA UNIVERSIDAD
DE ZARAGOZA

2016

25

Mohamed Taher Kahil

Water management under scarcity and climate change: methodological proposals and analysis of policy instruments

Departamento
Análisis Económico

Director/es
Albiac Murillo, José

ISSN 2254-7606

Prensas de la Universidad
Universidad Zaragoza

Servicio de Publicaciones

ISSN 2254-7606

Universidad Zaragoza

1542

Tesis Doctoral

WATER MANAGEMENT UNDER SCARCITY AND CLIMATE CHANGE: METHODOLOGICAL PROPOSALS AND ANALYSIS OF POLICY INSTRUMENTS

Autor

Mohamed Taher Kahil

Director/es

Albiac Murillo, José

UNIVERSIDAD DE ZARAGOZA

Análisis Económico

2015

UNIVERSIDAD DE ZARAGOZA

FACULTAD DE ECONOMÍA Y EMPRESA

DEPARTAMENTO DE ANÁLISIS ECONÓMICO

WATER MANAGEMENT UNDER SCARCITY AND CLIMATE CHANGE: METHODOLOGICAL PROPOSALS AND ANALYSIS OF POLICY INSTRUMENTS

Tesis Doctoral

Mohamed Taher Kahil

Zaragoza, Agosto 2015

UNIVERSIDAD DE ZARAGOZA
FACULTAD DE ECONOMÍA Y EMPRESA
DEPARTAMENTO DE ANÁLISIS ECONÓMICO

WATER MANAGEMENT UNDER SCARCITY AND CLIMATE CHANGE: METHODOLOGICAL PROPOSALS AND ANALYSIS OF POLICY INSTRUMENTS

Mohamed Taher Kahil

Tesis presentada para optar al grado de Doctor
por la Universidad de Zaragoza

Dirigida por el Doctor José Albiac Murillo

Zaragoza, Agosto 2015

“*If I am going to live below a dam I would much rather have it built by an engineer than an economist. Nevertheless, the economist comes into the picture perhaps by asking the awkward question as to whether the dam should have been built in the first place*” (Kenneth Boulding, 1964)

A mi familia

Agradecimientos

Durante estos cuatro años de formación como estudiante de doctorado, he tenido la oportunidad de trabajar y vivir en un aprendizaje continuo. Pero esto no hubiese sido posible sin la ayuda de muchas personas e instituciones, a quienes quiero expresar mi más sincero agradecimiento por el apoyo y la confianza que me han prestado de forma generosa.

Quiero empezar mis agradecimientos, como no puede ser de otra forma, con el Doctor José Albiac, el director de esta tesis. Ha sido y es para mí un privilegio poder trabajar bajo su dirección, por su confianza, su paciencia y ayuda constante, por todo el conocimiento que me ha transferido, por su disposición y plena dedicación, y por darme la oportunidad de viajar a muchos sitios maravillosos en búsqueda del aprendizaje. Es difícil sintetizar en pocas líneas todo mi agradecimiento, por lo tanto me limitaré a decir que si volviera a empezar la tesis otra vez, la volvería a hacer con él sin ninguna duda.

Mi siguiente agradecimiento va dirigido al Instituto Nacional de Investigación y Tecnología Agraria y Alimentaria (INIA), que me concedió una beca pre-doctoral para la formación de personal investigador en el Centro de Investigación y Tecnología Agroalimentaria de Aragón (CITA). Gracias al apoyo del INIA y el CITA, he podido continuar con mi formación, asistir a congresos y cursos, y realizar estancias al extranjero.

Quiero expresar mi agradecimiento también al Doctor Ariel Dinar, de la Universidad de California en Riverside (USA), por su confianza, su ayuda y su disponibilidad, y por su contribución importante en esta tesis. Al Doctor Jeff Connor, del CSIRO en Adelaida (Australia), por ayudarme a aprender nuevas herramientas que me han permitido aprender y ampliar mi campo de investigación. Al Doctor Frank Ward, de la Universidad de Nuevo Méjico en Las Cruces (USA), por su amabilidad, su paciencia, y por compartir conmigo su amplia experiencia en modelización hidro-económica.

En la parte técnica, esta tesis no se hubiera podido realizar sin los datos y los consejos que nos han facilitado muchos expertos, en especial Lorenzo Avella, Marta García-Molla, Carles Sanchis y Manuel Pulido-Velazquez (Universidad Politécnica de Valencia), Alfonso Calera, David Sanz y María Calera (Universidad de Castilla La Mancha), Eduardo Notivol, Sergio Lecina y Daniel Isidoro (CITA).

Gracias también a todos los compañeros y al personal de la Unidad de Economía Agraria y de los Recursos Naturales del CITA por facilitar mi estancia en dicha unidad y ayudarme de manera continua, en especial a Javier Tapia, Joaquín Moreno, Michael Bourne, Belinda López, Teresa Monzón y Encarna Esteban.

Quiero también dar las gracias a todos mis amigos por su amistad y su apoyo incondicional, en especial a Ahmed, Anouar, Alaeddine, Hedi, Mehdi, Mejdi, Mohamed, Nouredine, Ramzi, Talel, Taher, Wael y Walid.

Por último, no puedo terminar sin agradecer a mi querida familia, mis padres, Khadija y Jaloul, mi hermana Sabrine y mi novia Imen, por el apoyo incondicional que me han prestado, por aguantarme en todos los momentos difíciles por los que he pasado, y por aportarme la estabilidad emocional que he necesitado para realizar esta tesis. Esta tesis os la dedico especialmente a vosotros.

Contents

List of Tables

List of Figures

Chapter 1

General introduction

1.1 Background

Water is one of the most vital natural resources of the planet. Its importance stems from its role as a prerequisite for life, on the one hand, and its use as an input to economic production activities on the other. However, water demand has been increasing and continues to increase globally, as the world population grows and nations become wealthier. Water demands get closer and closer to the renewable freshwater resource availability, with water scarcity becoming a widespread problem during recent decades in most arid and semiarid regions around the world (Alcamo et al., 2007). There is a severe scarcity problem in almost all the important rivers in these regions, such as in the Nile, Ganges, Indus, Yellow, Yangtze, Tigris, Euphrates, Amu and Syr Darya, Murray-Darling, Colorado and Rio Grande (WWAP, 2006).

Water scarcity is created gradually by the decisions on water extractions in river basins linked to land use and economic activities. At first, water scarcity resulted from surface extractions, but recently it is worsening because of the unprecedented depletion of groundwater brought about by falling pumping costs. Between 1960 and 2000, groundwater extractions climbed from 310 to 730 km^3 per year pushing depletion up to 150 km^3 (Konikow, 2011). This staggering annual depletion amounts to 50 km^3 in the Indus-Ganges-Brahmaputra region, 24 km^3 in the USA, 13 km^3 in the Tigris-Euphrates region, and 9 km^3 in Northern China.

The problems arising from water scarcity could become critical because of climate change impacts. In some regions, climate change is expected to reduce freshwater supplies from surface and subsurface water resources, and also to increase of the recurrence, longevity and intensity of drought events. The reduction of water availability will be combined with increases of water demand for human and agricultural uses (IPCC, 2014). The combined effects of human-induced permanent water scarcity and climate change impacts portend unprecedented levels of water resources degradation.

The degradation of water resources is a threat to human water security and environmental biodiversity across the world, which so far has been addressed by large investments to ensure human security in medium and high income countries. However, the threats to natural ecosystems are hardly accounted for (Vörösmarty et al., 2010). Moreover, the most ideal sites for investments have been already used, leading to increased costs of new investments. These facts call for a reconsideration of the current water institutions and policies. The reason is not only to protect ecosystems, but also to substitute the escalating investments that ensure human security for more sustainable water management options.

The sustainable management of water is quite challenging because of the different types of goods and services provided by water. These goods and services can be classified as private goods, common pool resources, or public goods, depending on the degree of exclusion and rivalry in consumption.[1] Treated drinkable water in urban networks is close to a private good (rivalry and exclusion), water in surface watercourses and aquifers is close to a common pool resource (rivalry and non-exclusion), while water sustaining ecosystems comes close to a public good (non-rivalry and non-exclusion) (Booker et al., 2012). The management of water is governed by public policies because pure competitive markets fail to account for the common pool and public good characteristics of water.

One important question for future policy debates is the identification of potential water management policies to address scarcity, droughts and climate change. Suitable policies should improve economic efficiency, achieve environmental sustainability, and address equity. The evaluation of the effectiveness of water policies requires the development of analytical tools capable to integrate the key biophysical and socio-economic dimensions of water resources and to provide a better understanding of the spatial and inter-temporal effects of policy interventions.

1.2 Types of policy instruments

Economic theory describes three types of policy instruments that could be used to manage water resources. The first type is the "Pigou solution", which is based on taxation of water extractions (Pigou, 1920). This is the water pricing approach that is being implemented in the European water policies (EC, 2012). The second type is the "Coase solution", which is based on privatizing the resource and trading (Coase, 1960).

[1] A good is non-rival when consumption by one individual does not reduce consumption available to others (e.g. defense). A good is non-excludable when provision entails that its benefits are available to all (e.g. street lighting).

This is the water market approach that has been implemented in Australia. The third type is the common property governance, based on the evidence that coercive government rules fail because they lack legitimacy and knowledge of local conditions (Ostrom, 1990; Ostrom et al., 1999). This is the institutional approach, where stakeholders themselves design the rules and enforcement mechanisms for the sustainable management of common pool resources, although this approach has been mostly ignored by water policy decision makers and managers.

Mainstream water policies in some countries derive from the Dublin Statement on Water declaring water an economic good, and are based on so-called economic instruments such as water markets or water pricing. Besides the European Union and Australia, both water pricing and water markets are being considered at present for solving the acute water scarcity problems in China. These economic instruments 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 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.

1.3 Previous analysis of policy instruments

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; Scheierling 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).

The Murray-Darling Basin is at present involved in the most active water market in the world, and during the last drought this market has generated benefits of nearly 1 billion US dollars per year. A challenge to water markets is the third party effects such as environmental impacts, which would reduce the benefits of trading. Water markets reduce streamflows because previously unused water allocations are traded, and also because gains in irrigation efficiency at parcel level reduce return flows to the environment. This reduction in return flows has been analyzed both in Australia and the US (Qureshi et al., 2010; Howe et al., 1986). Another worrying effect is the large surge in groundwater extractions. It is estimated that groundwater extractions in the Murray-Darling Basin increased by 415 Mm^3 per year between 2000 and 2005. This represents a marked acceleration in previous historical rate of growth in groundwater extraction which grew by only 180 Mm^3 between 1984 and 2000. Other sources find substantially higher groundwater depletion in the Murray-Darling between 2002-2007 (last drought): extractions between 2002-2007 were 17000 Mm^3 per year, seven times above the allowed limits placed on groundwater users (2400 Mm^3), with additional groundwater depletion in the basin soaring up to 104 km^3 (Blewett, 2012).

Medellín et al. (2013) show the potential gains from water trading under droughts or climate change in California. The gains in the Central Valley of California are estimated at 1.4 billion US dollars. However, implementing these potential gains from trading is quite a challenge as the failure of the Water Bank experience in the 2009 drought shows: transfers were blocked by the water exporting regions and environmental NGOs. The attainment of this solution seems to require stronger institutions, involving stakeholders' cooperation.

Policies can also be irreversible. For example the "Tarifa 09", that subsidizes electricity to groundwater pumping for irrigation, was instituted in 1992 in Mexico to support Mexican farmers compete with USA and Canadian farmers within the North American Free Trade Agreement (NAFTA). Over the years that pervasive subsidy led to the destruction of more than 40 percent of major aquifers in Mexico. Efforts by the Government of Mexico to date to remove or reduce the subsidy have failed due to the strong opposition of the agricultural lobby (Muñoz et al., 2006).

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. Evidence on the importance of cooperation is provided by the case of the La Mancha aquifers (Esteban and Albiac, 2012). Subsidies to farmers and large public funds for buying water from farmers have failed to reduce extractions in the Western La Mancha aquifer, while extractions have been curbed in the Eastern La Mancha aquifer through stakeholders' cooperation within the basin authority. Carefully designed economic instruments have been used in the Eastern La Mancha aquifer to introduce more flexibility into the institutional process of decision making and implementation.[2]

1.4 Review of modeling approaches for water policy analysis

The complexity of water resource systems and the multiple challenges facing water stakeholders and policymakers have led to the development of new approaches to modeling water management and policies in the last three decades. Hydro-economic modeling has evolved from modeling only individual sector use at small spatial scales towards integrated modeling of a whole system of interrelated demand and supply elements. Integrated hydro-economic models are capable to efficiently and consistently account for the hydrologic, economic, institutional and environmental impacts of policy proposals to support the sustainable management of water resources (Booker et al., 2012).

Integrated hydro-economic models have been developed to solve different water management problems. Some examples of hydro-economic models application are the assessment of the potential of water markets to alleviate water scarcity in California (Vaux and Howitt, 1984), the modeling of intrastate and interstate markets for Colorado River water resources (Booker and Young, 1999), the analysis of the economic and environmental effects of limiting groundwater pumping in Texas (MacCarl et al., 1999), the development of an integrated hydrologic, agronomic and economic model for the Syr Darya basin in Central Asia (Cai et al., 2003), the assessment of the economic impacts of federal policy responses to drought in the Rio Grande basin (Ward et al., 2006), and the development of portfolios of water supply transfer in order to minimize the costs of meeting urban demand with a specified reliability in the Lower Rio Grande basin (Characklis et al., 2006).

Two main classifications of integrated hydro-economic models are presented in the literature based on how the hydrologic, economic, environmental and institutional components are integrated. The first one is 'compartment modeling' which treats the different components as separate sub-models, whose individual solutions are modified by some coordination methods. The alternative approach is 'holistic modeling' where

[2] A targeted public purchase of groundwater by the Jucar basin authority was in fact implemented during the last drought to confront the desiccation of the Jucar riverbed (CHJ, 2009).

the different components are integrated into a single consistent model, which is solved in its entirety, and information is transferred endogenously between the components. The endogenous treatment of the inter-relationships between the components allows a more effective combined environmental-economic analysis (Cai, 2008).

Holistic hydro-economic models have shown growing study interest and importance for integrated basin management. These models are typically developed as constrained optimization problems. Economic measures of the benefits and costs of water use are included in the objective function, while hydrologic and other factors are generally represented as constraints. Additional constraints are used to represent water institutions and environmental restrictions. Such models can be used for surface water dominated policy issues, groundwater issues, or physical environments in which combined ground and surface water use are important. Multiple supply and demand units and infrastructures could be represented, allowing for an explicit spatial analysis (Booker et al., 2012).

Despite the important achievements of previous studies, several gaps are not yet closed in the development of hydro-economic models. Booker et al. (2012) review the evolution of hydro-economic modeling and its capability for addressing system wide impacts. The authors indicate that hydro-economic modeling requires further advances in the dynamic and stochastic model dimensions, and also in the accurate understanding of interdependencies between the different model components. In addition, most existing hydro-economic models have been developed to find the best responses to climate and policy scenarios in basins. However, less attention has been paid to the strategic behavior of individual stakeholders, and their uptake of the basin-wide policy proposals.

1.5 Objectives of the research and methodology

The main objective of this research is the development and application of integrated hydro-economic models at basin scale. These models are used to analyze the impacts of climate change-induced water scarcity and droughts on the use of water in basins of arid and semiarid regions, and to assess the scope of options for achieving a sustainable management of these basins. The empirical findings of this research could assist policymakers in the design of sustainable water management policies at basin scale.

The thesis is divided into four articles (chapters 2 to 5) that present different hydro-economic modeling approaches addressing various climate scenarios and policy alternatives. The Jucar basin in Spain is chosen as the case study for the four chapters. This basin provides a good experimental field to apply hydro-economic modeling in arid and semiarid regions. The purpose is studying policies to confront water scarcity and the impact of droughts from the impending climate change. The Jucar region is semiarid and the river is under severe stress with acute water scarcity problems and escalating degradation of ecosystems. At present this is a common situation in many arid and semiarid basins around the world, including basins in Southern Europe, the Mediterranean region, Africa, the Middle East, Southwestern United States, and Australia. The findings in the Jucar could have important implications for water management in arid and semiarid basins around the world.

The first article (chapter 2) presents a hydro-economic model of the Jucar basin 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 is based on a detailed farm-level optimization of irrigation districts and an optimization of social surplus in urban water supply and demand. 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. This integrated model simulates demand nodes' behavior under different drought scenarios and policy intervention alternatives. 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 second article (chapter 3) develops a game theory framework in order to analyze cooperative water management policies that could address scarcity and drought in the Jucar basin. The existing literature, while assessing solutions to drought situations using integrated hydro-economic models, usually overlooks one important aspect, which is the strategic behavior of individual stakeholders. This gap is addressed in this paper by using several solution concepts and stability indexes from the cooperative game theory. Results demonstrate the importance of incorporating the strategic behavior of water stakeholders for the design of acceptable and stable basin-wide drought mitigation policies. The findings of this chapter provide clear evidence that achieving cooperation reduces drought damage costs. However, cooperation may have to be regulated by public agencies, such as a basin authority, when scarcity is very high, in order to protect ecosystems and maintain economic benefits.

The third article (chapter 4) presents a comparative analysis of the effectiveness of two popular incentive-based water management policies to address climate change impacts on irrigated agriculture in Southern Europe: water markets and irrigation subsidies. There are no studies in the European context providing a comparative analysis of the effectiveness of these two policies for irrigation adaptation to climate change, and the extent to which farmers could realize potential adaptation opportunities. The analysis is undertaken using a modeling framework that links hydrologic, agronomic, and economic variables within a discrete stochastic programming model. This model estimates farmers' responses to climate change and policy interventions in terms of long-run choices of capital investment in cropping and irrigation systems and short-run decision to irrigate or fallow land. The findings in this chapter highlight that climate change will likely have negative impacts on the irrigated agriculture and also on water-dependent ecosystems in Southern Europe. However, the severity of these impacts will depend on the degree of adaptation at farm level, the investment decisions by farmers, and the government policy choices.

The last article (chapter 5) presents a further enhancement of the hydro-economic model of the Jucar basin developed in chapter 1. This chapter includes the dynamic aspects into the hydro-economic modeling framework, following the recommendations by Booker et al. (2012) on the advances needed in hydro-economic modeling. This framework keeps track of all sources and demand nodes as water flows move from the headwaters to various downstream water uses. The framework integrates a spatially-explicit groundwater flow components. The methodological contribution to previous modeling efforts is the explicit specification of the aquifer-river interactions, which are important when aquifer systems make a sizable contribution to basin resources. This advanced framework is used for the assessment of different climate change scenarios and policy choices, looking for accurate evaluations of hydrologic, land use and economic outcomes. The results of this chapter provide valuable information on the basin scale climate change adaptation paths to guide alternative policy choices.

The contributions of this thesis relative to prior literature are both methodological and empirical. The hydro-economic modeling approaches presented here demonstrate the importance of integrated water resources modeling. This integration has considerable potential to support and advance environmental and economic policy assessments at basin scale, contributing towards the design of sustainable policies and management options. In addition, some methods are suggested in this research to fill the gaps not yet closed in the hydro-economic modeling literature. The gaps considered here are related first to the inclusion of the strategic behavior of individual stakeholders, and second to the dynamic and stochastic dimensions of the models. Empirically, the results of this thesis provide additional evidences on the advantages of stakeholders' cooperation for water management compared to other policy instruments, such as water markets and subsidies for investments in advanced irrigation technologies. The research provides also information on the required incentives and mechanisms to achieve a sustainable management of water resources in arid and semiarid basins under scarcity, droughts and climate change.

1.6 References

Alcamo, J., Florke, M., Marker, M., 2007. Future long-term changes in global water resources driven by socio-economic and climatic changes. *Hydrolog. Sci.* 52, 247-275.

Blewett, R., 2012. Shaping a Nation. A Geology of Australia. Geoscience Australia-ANU Press. Canberra.

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

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

Boulding, K., 1964. The economist and the engineer: Economic dynamics of water resources development. In Smith, E. and Castle, E. (Eds), Economics and public policy in water resources development. Iowa State University Press, Ames, pp. 82-92.

Cai, X., McKinney, D., Lasdon, L., 2003. Integrated hydrologic-agronomic-economic model for river basin management. *J. Water Res. Pl. Manag.* 129, 4-17.

Cai, X., 2008. Implementation of holistic water resources-economic optimization models for river basin management - Reflective experiences. *Environ. Modell. Softw.* 23, 2-18.

Characklis, G., Kirsch, B., Ramsey, J., Dillard, K., Kelley, C., 2006. Developing portfolios of water supply transfers. *Water Resour. Res.* 42, W05403, doi:10.1029/2005WR004424.

Coase, R., 1960. The problem of social cost. *J. Law Econ.* 3, 1-44.

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

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

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

European Commission (EC), 2012. A blueprint to safeguard Europe's water resources. European Commission. Brussels.

Hanemann, W., 1997. Price and rate structure. In Baumann, D., Boland, J, and Hanemann, W. (Eds), Urban Water Demand Management and Planning. McGraw-Hill, New York, pp. 137-139.

Howe, C., Schurmeier, D., Shaw, W., 1986. Innovative approaches to water allocation: The potential for water markets. *Water Resour. Res.* 22(4), 439-449.

Intergovernmental Panel on Climate Change (IPCC), 2014. Summary for Policymakers. In: Field, C., Barros, V., Dokken, D., et al. (Eds.), Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press. Cambridge.

Konikow, L., 2011. Contribution of global groundwater depletion since 1900 to sea level rise. *Geophys. Res. Lett.* 38, L17401.

McCarl, B., Dillion, C., Keplinger, K., Williams, R., 1999. Limiting pumping from the Edwards Aquifer: an economic investigation of proposal, water markets, and spring flow guarantees. *Water Resour. Res.* 34(4), 1257-1268.

Medellín, J., Howitt, R., Lund, J., 2013. Modeling Economic-Engineering Responses to Drought: The California Case, In Schwabe, K., Albiac, J., Connor, J., Hassan, R., Meza, L. (Eds), Drought in Arid and Semi-Arid Environments: A Multi-Disciplinary and Cross-Country Perspective, Springer, Dordrecht, pp. 341-356.

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

Muñoz, C., Avila, S., Jaramillo, L., Sainz, J., Martinez, A., 2006. Agriculture demand for groundwater in Mexico: Impact of water right enforcement and electricity user-fee on groundwater level and quality. Working paper. Instituto Nacional de Ecología, Mexico.

Ostrom, E., 1990. Governing the commons: The evolution of institutions for collective action. Cambridge University Press. Cambridge.

Ostrom, E., Burger, J., Field, C., Norgaard, R., Policansky, D., 1999. Revisiting the commons: local lessons, global challenges. *Science* 284, 278-282.

Pigou, A., 1920. The Economics of Welfare. Macmillan. New York.

Qureshi, M., Schwabe, K., Connor, J., Kirby, M., 2010. Environmental water incentive policy and return flows. *Water Resour. Res.* 46(4), 1-12.

Scheierling, S., Young, R., Cardon, G., 2004. Determining the price responsiveness of demands for irrigation water deliveries versus consumptive use. *J. Agr. Resour. Econ.* 29 (2), 328-345.

Schwabe K., Albiac, J., Andreu, J., Ayers, J., Caiola, N., Hayman, P., Ibañez, C., 2013. Summaries and Considerations. In Schwabe, K., Albiac, J., Connor, J., Hassan, R., Meza, L (Eds), Drought in Arid and Semi-Arid Environments: A Multi-Disciplinary and Cross-Country Perspective, Springer, Dordrecht, pp. 471-507.

Vaux, H., Howitt, R., 1984. Managing water scarcity: An evaluation of interregional transfers. *Water Resour. Res.* 20 (7), 785-792.

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

Ward, F., Hurd, B., Rahmani, T., Gollehon, N., 2006. Economic impacts of federal policy responses to drought in the Rio Grande Basin. *Water Resour. Res.* 42, W03420, doi:10.1029/2005WR004427.

World Water Assessment Programme (WWAP), 2006. Water: A shared responsibility. UNESCO-Berghahn. New York.

Chapter 2

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

Abstract

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.

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

2.1 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), Pulido-Velazquez 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 towards a more sustainable water management.

2.2 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 2.3).

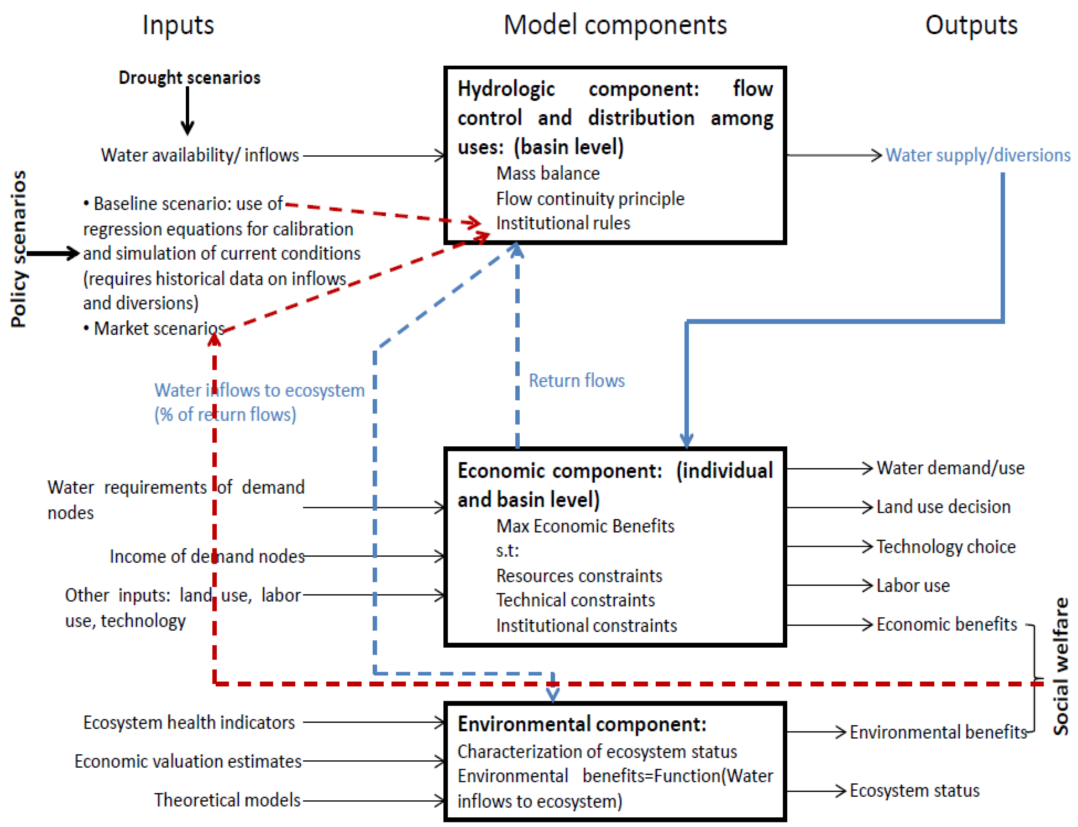
The flows of water are routed between nodes using basic hydrologic concepts, such as mass balance and river flow continuity equations. The mass balance principle could be applied for surface flow, reservoir, and aquifer levels. The model is initially constrained by a known volume of water availability 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).

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

Figure 2.1. Modeling framework.



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.

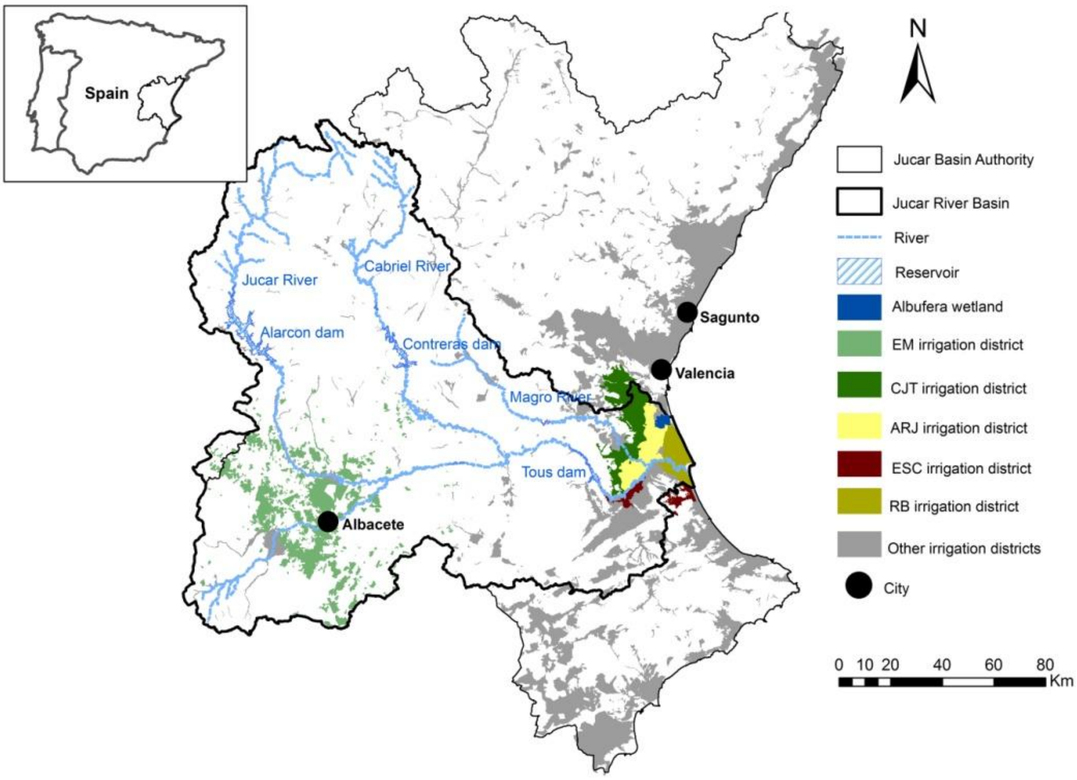
The modeling framework described in this section is summarized in figure 2.1 and it is applied to the drought management problem in an arid and semiarid basin in Southeastern Spain, the Jucar River Basin. The next section provides background information on the basin, and the following sections present the design and calibration of the reduced form hydrological model and that of the economic models to the conditions in the Jucar River Basin.

2.3 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

Figure 2.2. Map of the Jucar River Basin.



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 2.2). 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.

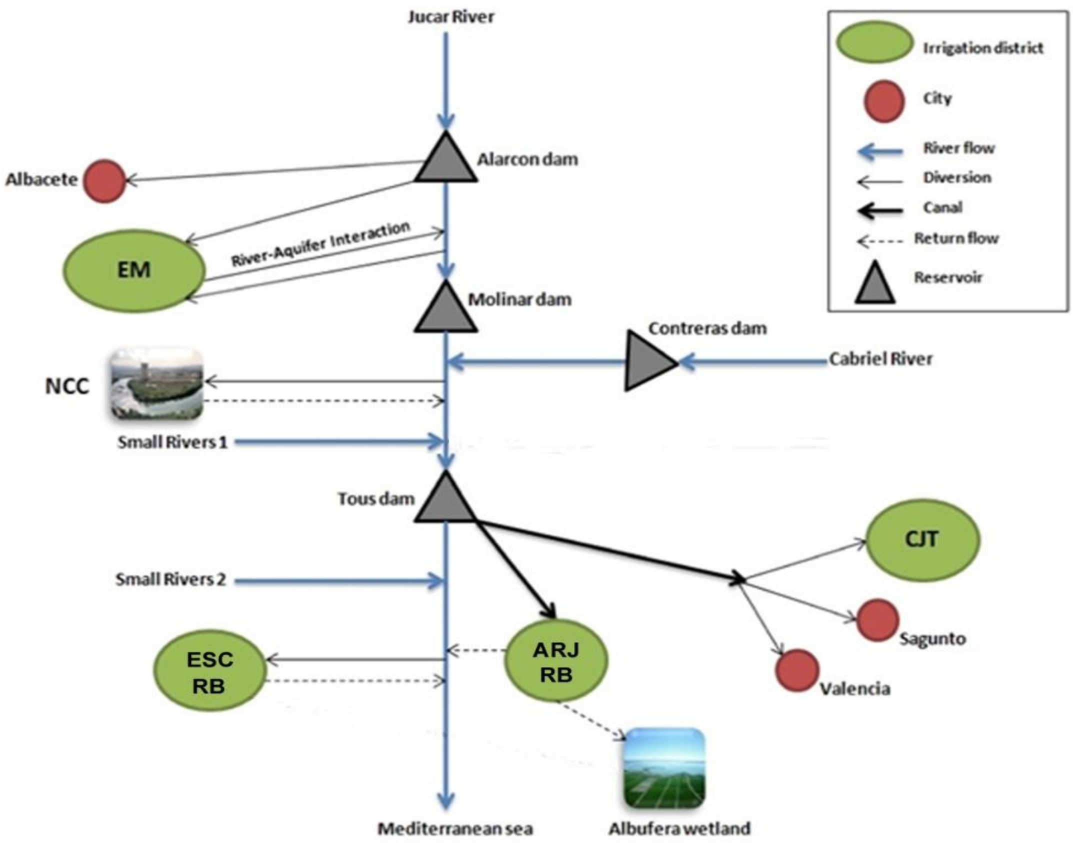
At present, renewable water resources in the JRB are nearly 1,700 Mm^3, of which 930 are surface water and 770 are groundwater resources. Water extractions are 1,680 Mm^3, very close to renewable resources, making the JRB an almost closed water system. Extractions for irrigated agriculture are nearly 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).

Figure 2.3. Jucar River Basin network.



The Albufera wetland is a freshwater lagoon with an area covering 2,430 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).

2.4 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.

2.4.1 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 2.3 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:

$$Wout_d = Win_d - Wloss_d - Div_d^{IR} - Div_d^{URB} \tag{2.1}$$

$$Win_{d+1} = Wout_d + r_d^{IR} \cdot (Div_d^{IR}) + r_d^{URB} \cdot (Div_d^{URB}) + RO_{d+1} \tag{2.2}$$

$$Wout_d \geq E_d^{min} \tag{2.3}$$

The mass balance equation (2.1) 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 (2.2) 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 (2.3) 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 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 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.

2.4.2 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:

$$Max\ B_k^{IR} = \sum_{ij} C_{ijk}^{'} \cdot X_{ijk} \tag{2.4}$$

subject to

$$\sum_i X_{ijk} \leq Tland_{kj}\ \ ;\ \ j = flood, sprinkler, drip \tag{2.5}$$

$$\sum_{ij} W_{ijk} \cdot X_{ijk} \leq Twater_k \tag{2.6}$$

$$\sum_{ij} L_{ijk} \cdot X_{ijk} \leq Tlabor_k \tag{2.7}$$

$$X_{ijk} = \sum_n \alpha_n \cdot X_{ijkn}\ ;\ \sum_n \alpha_n = 1\ ;\ \alpha_n \geq 0 \tag{2.8}$$

$$X_{ijk} \geq 0 \tag{2.9}$$

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 (2.5) represents the irrigation area equipped with technology j in district k, $Tland_{kj}$. The water constraint (2.6) 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 (2.7) 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 (2.8) 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 2.1).

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:

$$Max\ B_u^{URB} = \left(a_{du} \cdot Q_{du} - \frac{1}{2} \cdot b_{du} \cdot Q_{du}^2 - a_{su} \cdot Q_{su} - \frac{1}{2} \cdot b_{su} \cdot Q_{su}^2\right) \quad (2.10)$$

subject to

$$Q_{du} - Q_{su} \leq 0 \quad (2.11)$$

$$Q_{du}\ ;\ Q_{su} \geq 0 \quad (2.12)$$

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 (2.11) 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 2.1).

2.4.3 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:

$$E_{Albufera} = \alpha \cdot r_{ARJ}^{IR} \cdot \left(Div_{ARJ}^{IR}\right) + \beta \cdot r_{RB}^{IR} \cdot \left(Div_{RB}^{IR}\right) \tag{2.13}$$

$$B_{Albufera} = \begin{cases} \delta_1 & if\ 0 \leq E_{Albufera} \leq E_1 \\ \delta_2 + \rho_2 \cdot E_{Albufera} & if\ E_1 < E_{Albufera} \leq E_2 \\ \delta_3 + \rho_3 \cdot E_{Albufera} & if\ E_2 < E_{Albufera} \leq E_3 \end{cases} \tag{2.14}$$

where equation (2.13) 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 (2.14) 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

Table 2.1. 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

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 2.1.

2.4.4 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:

$$Max\left(\sum_l B_l + B_{Albufera}\right) \quad \forall\, l = k, u \tag{2.15}$$

subject to the constraints in equations (2.1), (2.2), (2.3), (2.5), (2.6), (2.7), (2.8), (2.9), (2.11), (2.12), (2.13), 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:

$$Div_d^l = f(Win_d) \quad \forall\, l, d \tag{2.16}$$

$$\sum_{ld} Div_d^l \le \overline{W} \tag{2.17}$$

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 (2.16) and (2.17) are used to allocate water among users under the baseline policy (institutional approach). Equation (2.16) 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. Equation (2.17) 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 (equation 2.15), subject to the total basin water availability (equation 2.17). The regression equations (equation 2.16) 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 (2.7) 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.

2.4.5 Model application and scenarios

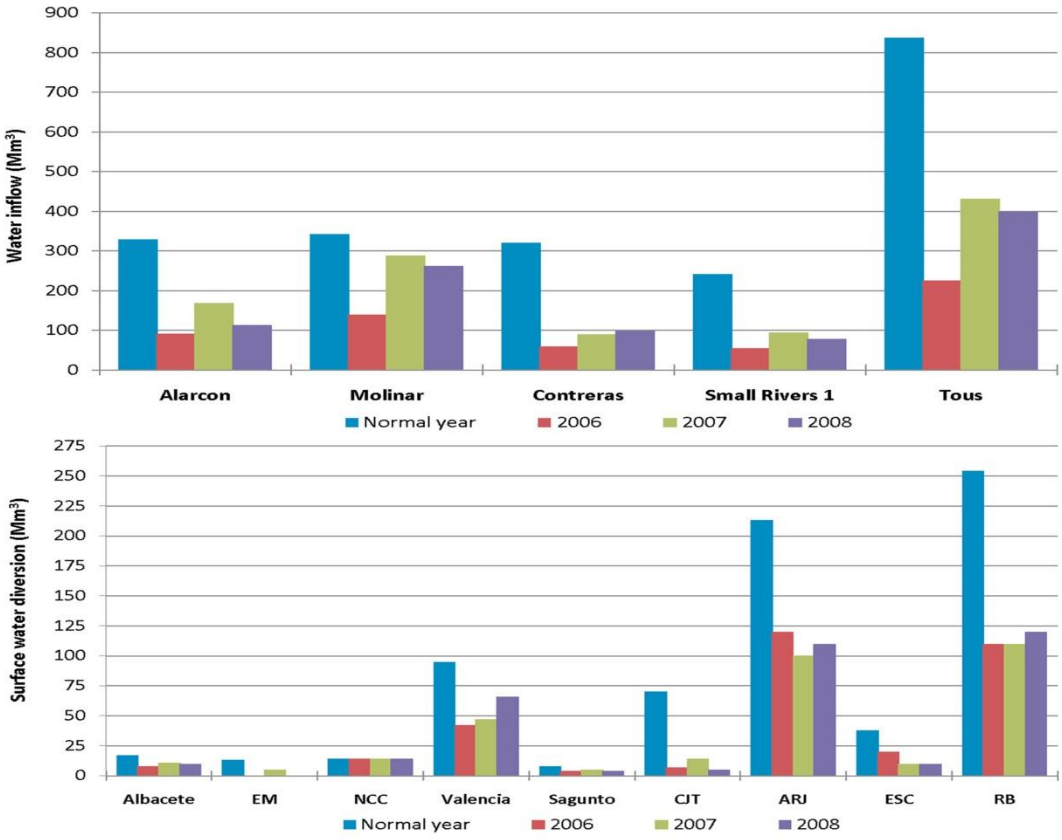
The modeling framework is used to analyze the impacts of climate change-induced drought on water uses in the JRB. Given the uncertainty associated with future climate change, three alternate drought scenarios are developed to reflect a range of possible future water availability in the basin. Drought scenarios expressed as a percentage reduction of normal year water inflows are the following: mild (-22 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:

***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.

Figure 2.4. Surface water inflows to the main reservoirs and river reaches (top) and diversions for the demand nodes (down) in the Jucar River Basin.



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.

Table 2.2. 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 $p<0.2$.

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.

2.5 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) (Figure 2.4).

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 (Figure 2.4).

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

Table 2.3. Comparison between simulated (Sim) and observed (Ob) water diversions (Mm^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**

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 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 2.2 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 2.3).

2.6 Results and discussion

The economic and environmental outcomes from the three policy alternatives and drought scenarios are depicted in tables 2.4, 2.5, and 2.6.

2.6.1 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 2.4). Water use is 1,149 Mm^3, of which 672 is surface water and 477 is groundwater resources (Table 2.5). Irrigation activities generate 190 million € from using 1,030 Mm^3. The social surplus of urban centers is 283 million € and they use 119 Mm^3. About 60 Mm^3 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 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 296 Mm^3 and groundwater extractions up to 52 Mm^3. 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.

Table 2.4. 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 (10⁶ €)				
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 (10⁶ €)	74.7	37.2	33.0	33.0
Social welfare (10⁶ €)	**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 (10⁶ €)				
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 (10⁶ €)	74.7	33.0	33.0	33.0
Social welfare (10⁶ €)	**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 (10⁶ €)				
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 (10⁶ €)	277.6	275.9	272.6	255.7
Social welfare (10⁶ €)	**755.6**	**732.4**	**704.2**	**656.7**
Irrigation labor use (Jobs)	14,610	13,720	12,440	10,560

* 1 job unit= 1,920 hours/year.

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 2.6). 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.

Table 2.5. Water use and return flows under the policy and drought scenarios (Mm^3).

Aggregate results	Normal flow	Mild drought	Severe drought	Very severe drought
		Baseline policy		
Water use				
Irrigation sector	1,030	908	793	683
Urban sector*	119	105	90	74
Total	**1,149**	**1,013**	**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	1,030	908	793	683
Urban sector	119	105	90	74
Total	**1,149**	**1,013**	**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	**1,055**	**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.

The cropping pattern and irrigation technology distribution 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, 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_1

Table 2.6. 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

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 50 percent 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 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 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.

2.6.2 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 percent compared to the *Baseline policy*. Irrigation benefits increase under water markets up to 9 percent, 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 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 *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

Mm^3 (10%) compared to the *Baseline* 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 *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 Mm^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.

2.6.3 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 percent 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 percent 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 Mm^3 under normal flow scenario to 201 Mm^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 Mm^3, securing always a fixed amount of water (138 Mm^3) flowing to the wetland. This amount is well above the minimum environmental requirements of the Albufera wetland set by the basin authority (60 Mm^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 Mm^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 (Table 2.7). 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

Table 2.7. Sensitivity analysis with different ecosystem values.

	Normal flow	Mild drought	Severe drought	Very severe drought
Base case ecosystem value (13,600 €/ha)				
Irrigation private benefits (10^6 €)	195.4	180.2	165.2	160.1
Environmental benefits (10^6 €)	277.6	275.9	272.6	255.7
Inflows to Albufera from trade (Mm^3)	89	100	115	131
High ecosystem value (340,000 €/ha)				
Irrigation private benefits (10^6 €)	195.4	180.2	165.2	160.1
Environmental benefits (10^6 €)	7281.9	7280.2	7276.8	7260.0
Inflows to Albufera from trade (Mm^3)	89	100	115	131
Low ecosystem value (544 €/ha)				
Irrigation private benefits (10^6 €)	191.6	176.3	163.1	147.5
Environmental benefits (10^6 €)	3.2	1.6	0.0	1.3
Inflows to Albufera from trade (Mm^3)	21	33	45	0

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.

2.7 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.

2.8 References

Alcamo, J., Henrichs, T., Rösch, T., 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.

Andreu, J., Capilla, J., Sanchis, E., 1996. AQUATOOL, a generalized decision-support system for water-resources planning and operational management. *J. Hydrol.* 177 (3-4), 269-291.

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

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

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

Cai, X., McKinney, D., Lasdon, L., 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 Confederación Hidrográfica 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.

Connor, J., Kaczan, D., 2013. Principles for economically efficient and environmentally sustainable water markets: The Australian experience. In Schwabe, K., Albiac, J., Connor, J., Hassan, R., Meza, L. (Eds), Drought in Arid and Semi-Arid Environments: A Multi-Disciplinary and Cross-Country Perspective, Springer, Dordrecht, pp. 357-374.

Cornish, G., Bosworth, B., Perry, C., Burke, J., 2004. Water charging in irrigated agriculture: An analysis of international experience, FAO water report 28, FAO, Rome.

Del Saz, S., Perez L., 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., Albiac J., 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.

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

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

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., Pulido, M., Rosenberg, D., Medellín, J., Lund, J., Howitt, R., 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., Xu, F., Marques, J., Salas, F., 2010. Application of indicators for the assessment of ecosystem health, In Handbook of Ecological Indicators for Assessment of Ecosystem Health, edited by Jorgensen, S., Xu, F., Costanza, R., pp 9-75, CRC press, Boca Raton.

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

Kirby, M., Bark, R., Connor, J., Qureshi, E., Keyworth, S., 2014. Sustainable irrigation: How did irrigated agriculture in Australia's Murray–Darling Basin adapt in the Millennium Drought?. *Agr. Water Manage* 145, 154-162.

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

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

Maneta, M., Torres, M., Wallender, W., Vosti, S., Howitt, R., Rodrigues, L., Bassoi, L., Panday, S., 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.

McKinney, D., Cai, X., Rosegrant, M., Ringler, C., Scott, C., 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.

Molinos, M., Hernández, F., Mocholí, M., Sala, R., 2014. A management and optimisation model for water supply planning in water deficit areas. *J. Hydrol.* 515, 139-146.

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., McCarl, B., 1991. Exact aggregation in mathematical programming sector models. *Can. J. Agr. Econ.* 39, 319-334.

Ostrom, E., Janssen, M., Anderies, J., 2007. Going beyond panaceas. PNAS 104 (39), 15176-15178.

Pulido-Velazquez, M., Andreu, J., Sahuquillo, A., Pulido, D., 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., Castaño, S., Cassiraga, E., Sahuquillo, A., Gómez, J., Peña, S., Calera, A., 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., Carpenter, S., Foley, J., Walker B., 2001. Catastrophic shifts in ecosystems. *Nature* 413, 591-596.

Schwabe, K., Albiac, J., Connor, J., Hassan, R., Meza, L., (Eds), 2013. Drought in arid and semi-arid regions: A multi-disciplinary and cross-country perspective, Springer, Dordrecht.

Varela, C., Blanco, I., Swartz, C., Downing, T., 2011. Balancing groundwater conservation and rural livelihoods under water and climate uncertainties: An integrated hydro-economic modeling framework. *Global Environ. Chang.* 21, 604-619.

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

Ward, F., 2014. Economic impacts on irrigated agriculture of water conservation programs in drought. *J. Hydrol.* 508, 114-127.

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

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

Chapter 3

Cooperative water management and ecosystem protection under scarcity and drought in arid and semiarid regions

Abstract

Climate change impacts and the growing concern on environmental water demand are further increasing competition for scarce water resources in many arid and semiarid regions worldwide. Under these circumstances, new water allocation mechanisms based on the involvement of stakeholders are needed, for an efficient and fair allocation of water and income among uses. This paper develops a cooperative game theory framework in order to analyze water management policies that could address scarcity and drought in a typical arid and semiarid basin in Southeastern Spain. The results provide clear evidence that achieving cooperation reduces drought damage costs. However, cooperation may have to be regulated by public agencies, such as a basin authority, when scarcity is very high, in order to protect ecosystems and maintain economic benefits. The cooperative game theory solutions and stability indexes examined in this paper demonstrate the importance of incorporating the strategic behavior of water stakeholders for the design of acceptable and stable basin-wide drought mitigation policies.

Keywords Cooperative solutions, Game theory, River basin modeling, Water economics, Water scarcity

3.1 Introduction

Global water resources are under increasing pressures that create growing water scarcity and quality problems, giving rise to complex social conflicts and environmental degradation. Water extractions across the world have increased more than six fold in the last century, much above the rate of population growth (UNDP, 2006). It is estimated that about 35 percent of the world population suffers from severe water stress and 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).

Water scarcity has become widespread in most arid and semiarid regions, including river basins such as the Yellow, Jordan, Murray-Darling, Colorado, and Rio Grande (UNDP, 2006; Schwabe et al., 2013). Projected future climate change impacts would further exacerbate the current situation of water scarcity in arid and semiarid regions. These regions would likely suffer a decrease in water resources availability and experience longer, more severe, and frequent droughts (IPCC, 2014).

Emerging social demands for environmental protection in the form of secured minimum flows for water-dependent ecosystems further increase competition for already scarce water in arid and semiarid regions, especially during dry years. Water-dependent ecosystems, such as wetlands, provide a diverse range of goods and services to society, including habitat for valuable species, flood control, groundwater replenishment, water quality improvement, waste disposal, and recreational opportunities (Woodward and Wui, 2001). However, water-dependent ecosystem services are external to markets, and their social values are overlooked in water allocation decisions. For instance, an estimated 50 percent of world wetlands have disappeared over the last century (Finlayson et al., 1999).

Several policy responses have been suggested to cope with water scarcity and to mitigate the negative impacts of droughts for the different water use sectors. These policies include reducing water allocations, water transfers, conjunctive use of ground and surface waters, groundwater banking, recycling and reuse of wastewater, seawater desalination, improving water use efficiency, adopting water conserving-technologies, changing crop mix, setting minimum environmental flows, and implementing economic instruments such as water pricing and water trade including water purchases for environmental purposes.

These policy alternatives have been previously analyzed in several studies such as Booker et al. (2005); Howitt et al. (2014); Kirby et al. (2014); and Zilberman et al. (1998). However, the existing literature, while assessing solutions to drought situations using engineering, economic and institutional approaches, usually overlooks one important aspect, which is the strategic behavior of individual stakeholders. The analysis of the strategic behavior of stakeholders is essential to test the acceptability and stability of policy solutions aimed at basin-wide drought mitigation.

This gap is addressed in this paper by developing a cooperative game theory (CGT) framework in order to analyze water management policies to deal with scarcity and drought at basin scale. The paper contributes to the literature on water policy through the inclusion of the strategic behavior of various stakeholders, and ecosystem benefits in the river water management problem. Several CGT solution concepts and stability indexes are used in order to find efficient and fair allocations of water and income among river users under various climate scenarios. In addition, the analysis considers the likelihood for ecosystem protection success.

The CGT deals with games in which stakeholders (players) choose to cooperate by forming coalitions and sharing fairly the benefits from those coalitions. In particular, CGT favors agreements that include all possible players (grand coalition) and it provides several benefit sharing mechanisms (solution concepts) based on different notions of fairness. The purpose is finding the incentives for cooperation among stakeholders in order to achieve the economic efficient outcomes for the coalitions. The advantage of using CGT compared to conventional optimization models is its ability to address both efficiency and equity principles, which would promote acceptable and stable cooperative outcomes (Dinar et al., 2008).

The CGT has been applied to different water management problems in the literature (Parrachino et al., 2006; Madani, 2010). Some examples are the allocation of the costs of a multipurpose water resources development project in the Tennessee Valley Authority (Straffin and Heaney, 1981), the allocation of environmental control costs among polluters in the San Joaquin Valley of California (Dinar and Howitt, 1997), the equitable distribution of benefits among competing water uses at basin scale in Canada (Wang et al., 2008), the efficient sharing of a hypothetical river among countries (Ambec and Ehlers, 2008), the allocation of the benefits of cooperative groundwater management among pumpers in the Eastern La Mancha aquifer in Spain

(Esteban and Dinar, 2013), and the development of optimal operation policies for a hypothetical multi-operator reservoir systems (Madani and Hooshyar, 2014).

The CGT framework is applied to the Jucar River Basin (JRB) of Spain, which is a good case for studying the strategic behavior of stakeholders and policies to confront water scarcity and drought impacts from the impending climate change. The JRB region is semiarid and the river is under severe stress with acute water scarcity problems and escalating degradation of ecosystems. Another interesting aspect of the JRB is that there have been already successful policies leading to stakeholders' cooperation. In particular, the curtailment of water extractions in the Eastern La Mancha aquifer that were threatening the activities of downstream stakeholders (Esteban and Albiac, 2012).

3.2 Cooperative game theory framework

This section presents the CGT framework used to analyze water management policies addressing scarcity and drought at basin scale. Assume that a basin includes $n>1$ users (players in the game). The users consider a cooperative management of the basin by agreeing to share water resources. Initially, the users have predetermined administrative water allocations depending on the climate condition. Under the cooperative water sharing agreement, the agency responsible for water allocation reallocates water among uses so that the whole basin benefits are maximized. When additional benefits are obtained through this cooperative agreement compared to non-cooperation (status quo), the water agency needs to distribute these benefits among the cooperating users in a fair way that would sustain cooperation.

Let N be the set of all players in the game, S is the set of all feasible coalitions, and s ($s \in S$) is one feasible coalition. The singleton coalitions are $\{l\}$, $l=1,2,...,n$, and the grand coalition is $\{N\}$. Assume that the objective of the water agency is to maximize the benefits, f^s, of any feasible coalition in the basin, s, by efficiently allocating water among the players in that coalition. Let $\nu(s)$ be the characteristic function of coalition s, which is the best value that such coalition can obtain. The cooperative water sharing agreement takes the following form:

$$\nu(s) = Max\ f^s = \sum_{l \in s} B_l \tag{3.1}$$

subject to

$$\sum_{l \in s} WU_l \leq WA_s \tag{3.2}$$

where B_l is the private net benefits from water use of player l in coalition s. The water constraint (3.2) states that the sum over players, l, in coalition, s, of water use by each player, WU_l, must be less than or equal to water available for that coalition, WA_s.

When additional benefits are obtained through this cooperative agreement compared to non-cooperation, the water agency overseeing the agreement needs to allocate these benefits among the cooperating players in a fair way in order to secure the acceptability and stability of the agreement. These allocations could be determined using the CGT solution concepts. A necessary condition for cooperation in the basin is that the benefits obtained by each cooperating player under full cooperation (grand coalition) are greater than what each player can obtain under non-cooperation (singleton coalition), or by participating in partial cooperative arrangements (partial coalitions).

Let Ω_l^a be the allocated cooperative benefit (payoff) to player l using the CGT solution concept, a. A feasible cooperative allocation should satisfy the following three requirements:

$$\Omega_l^a \geq \nu(\{l\}) \qquad \forall l \in N \tag{3.3}$$

$$\sum_{l\in s} \Omega_l^a \geq \nu(s) \quad \forall s \in S \tag{3.4}$$

$$\sum_{l\in N} \Omega_l^a = \nu(N) \tag{3.5}$$

Equation (3.3) fulfills the condition for individual rationality, which means that the allocated benefits from full cooperation to player l, Ω_l^a, must be greater than or equal to its benefits from non-cooperation, $\nu(\{l\})$. Equation (3.4) fulfills the group rationality condition, which means that the sum of full cooperative benefit allocations to any group of players, $\sum_{l\in s} \Omega_l^a$, must be greater than or equal to the total obtainable benefits under any coalition s that includes the same players, $\nu(s)$. Equation (3.5) fulfills the efficiency condition, which means that the total obtainable benefits under the grand coalition, $\nu(N)$, must be allocated to the members of that coalition, $\sum_{l\in N} \Omega_l^a$.

An allocation that satisfies these three requirements is in the Core of the cooperative game (Gillies, 1959). The Core is a set of game allocation gains that is not dominated by any other allocation set. The Core provides information about the range of acceptable solutions for each player and allows for ranking the players' preferences over the possible cooperative solutions. Satisfying the Core conditions for a cooperative solution is a necessary condition for its acceptability by the players. Therefore, solutions not included in the Core are not acceptable and not stable (Shapley, 1971).

Three CGT solution concepts based on different notions of fairness are used in this paper to allocate the gains from cooperation among the players: the Shapley value, the Nash-Harsanyi, and the Nucleolus.

The Shapley value allocates Ω_l^{Sh} to each player based on the weighted average of their contributions to all possible coalitions. The Shapley value is based on the intuition that the allocation that each player receives should be proportional to his contribution. Players who add nothing, should receive nothing and players who are indispensable should be allocated a lot (Shapley, 1953). The Shapley solution takes the following form:

$$\Omega_l^{Sh} = \sum_{\substack{s \in S \\ l \in s}} \frac{(n-|s|)!(|s|-1)!}{n!} \cdot \big(\nu(s) - \nu(s-\{l\})\big) \quad \forall l \in N \tag{3.6}$$

where n is the total number of players in the game, $|s|$ is the number of players participating in coalition s, and $\nu(s-\{l\})$ is the value of coalition s without member l.

The Nash–Harsanyi solution (Harsanyi, 1959) to an n-person bargaining game is a modification to the two-player Nash solution (Nash, 1953). This solution provides an allocation to each player, Ω_l^{NH}, by maximizing the product of the incremental gain of the players from cooperation. The Nash-Harsanyi solution takes the following form:

$$Max \ \prod_{l \in N} (\Omega_l^{NH} - \nu(\{l\})) \tag{3.7}$$

subject to the Core conditions (equations (3.3) to (3.5)). The Nash-Harsanyi solution is unique and it is in the Core (if it is not empty).

The Core of a cooperative game in the characteristic function form may be empty because certain partial coalitions provide greater payoff than the grand coalition. Conversely, conditions may arise where the Core does exist but is too large and leaves the allocation problem open for further bargaining. The Nucleolus solves this problem by minimizing the worst inequity or dissatisfaction of the most dissatisfied coalition (Schmeidler, 1969). The Nucleolus of the benefit allocation game can be determined by finding ε through the following optimization model:

$$Max \ \varepsilon \tag{3.8}$$

subject to

$$\varepsilon \leq \sum_{l \in s} \Omega_l^{Nu} - \nu(s) \quad \forall s \in S \tag{3.9}$$

$$\sum_{l \in N} \Omega_l^{Nu} = \nu(N) \tag{3.10}$$

$$\varepsilon \lesseqgtr 0 \tag{3.11}$$

where ε is the maximum tax imposed on or subsidy provided to all coalitions to keep them in the Core. The Nucleolus allocation, Ω_l^{Nu}, is a single solution that is always in the Core, if the Core is not empty.

The fulfillment of the Core requirements for a CGT allocation solution is a necessary condition for its acceptability by the players. However, being in the Core does not guarantee the stability for a solution, as some players may find it relatively unfair compared to other solutions. The consequence is that some players might threaten to leave the grand coalition and form partial coalitions because of their critical position in the grand coalition (Dinar and Howitt, 1997). The stability of any solution is important given the existence of considerable fixed investments and transaction costs, so that a more stable solution might be preferred even if it is harder to implement.

Some methods are suggested in the literature to evaluate the stability of the CGT allocation solutions (Dinar and Howitt, 1997). For instance, Loehman et al. (1979) used an ex-post approach to measure power in a cooperative game. This approach is similar to the one suggested by Shapley and Shubik (1954) for measuring power in voting games. The Loehman power index (θ_l^a) compares the gains to a player with the gains to the coalition. The power index is the following:

$$\theta_l^a = \frac{\Omega_l^a - v(\{l\})}{\sum_{l\in N}\left(\Omega_l^a - v(\{l\})\right)} \ , \ \ \sum_{l\in N}\theta_l^a = 1\, , \ \ a = Sh, NH, Nu \tag{3.12}$$

where Ω_l^a is the allocation solution for player l using the CGT solution concept a. The power index of each player is used as an indicator of the stability of the allocation solution. The higher the power index of a player, the higher that player's propensity for cooperating and staying in the grand coalition. If the power is distributed more or less equally among the players, then the coalition is more likely to be stable. The coefficient of variation of the power indexes of the different players for an allocation solution is defined as the stability index of the grand coalition $\overline{\theta_a}$. The greater the value of $\overline{\theta_a}$ the larger the instability of the allocation solution.

The theoretical CGT framework proposed in this section is applied to the water management problem in the JRB. The next section describes the empirical river basin model of the JRB that is used to calculate the value of the characteristic function of various coalitional arrangements.

3.3 Empirical river basin model

The empirical river basin model includes the main users in the JRB: irrigation activities, urban uses, and aquatic ecosystems needs. A specific model for optimizing each and all water use sectors has been built, and these models are linked, using a reduced form hydrological model developed and calibrated to the JRB conditions in chapter 2 of this thesis.

3.3.1 Study area

The JRB is located in the regions of Valencia and Castilla La Mancha in Southeastern Spain and it extends over 22,400 km^2. Renewable water resources in the JRB are nearly 1,700 Mm^3. Water extractions are 1,680 Mm^3, very close to renewable resources, making the JRB an almost closed water system (CHJ, 2009).

Extractions for irrigated agriculture are about 1,400 Mm^3 per year, which represent 84 percent of total water extractions, to irrigate 190,000 ha. The major irrigation districts are: the Eastern La Mancha aquifer district (EM) in the upper Jucar, the traditional districts of Acequia Real del Jucar (ARJ), Escalona y Carcagente (ESC) and Ribera Baja (RB) in the lower Jucar, and the the Canal Jucar-Turia district (CJT) situated in the adjacent Turia River Basin. Urban and industrial extractions are about 270 Mm^3, serving more than one million inhabitants located mostly in the cities of Valencia, Sagunto and Albacete (CHJ, 2009).

Expansions of water extractions in the basin and the severe drought spells in recent decades have triggered considerable negative environmental and economic impacts. 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. There have been also negative impacts on downstream water users, where water availability has been reduced substantially in the last forty years. Consequently, the dwindling irrigation return flows in the lower Jucar have caused serious environmental problems to the Albufera wetland, the main aquatic ecosystem in the JRB, which is mainly fed by these return flows (Garcia-Molla, 2013).

The Albufera wetland is a freshwater lagoon with an area covering 2,430 ha, supporting very rich aquatic ecosystems. Since 1989, the Albufera was included in the list of wetlands of international importance, and was declared a special protected area for birds. The Albufera receives water from the return flows of irrigation in the lower Jucar, mainly from the ARJ and the RB districts. Other flows originate from the Turia River Basin, and from discharge of untreated and treated urban and industrial wastewaters. Currently, the Albufera wetland suffers from reduction of inflows and the degradation of their quality. These problems are driven by the reduced flows originating from the Jucar River, and by deficiencies in the sewage disposal and treatment systems from adjacent municipalities, causing severe damages to the Albufera wetland, such as the loss of biodiversity, the decrease in recreation services, and the decline of fishing activities (Sanchis, 2011).

3.3.2 The model

The hydro-economic model of the JRB integrates hydrologic, economic, environmental, and institutional variables within a single framework. The model accounts for decision processes made by irrigators in the five major districts (EM, CJT, ARJ, ESC, and RB) and by urban users in the three major cities (Valencia, Albacete, and Sagunto) in the basin. In addition, the model includes environmental benefits provided by the Albufera wetland to society. Numerous small demand units in the basin are not included in the model. The model runs on an annual basis, and its main focus is on the allocation and utilization of surface water. Groundwater use and management are not taken into account in this paper.

In order to link the different components of the river basin model and to simulate the spatial impact of drought in the JRB, a reduced form of the hydrological model of the basin is used (CHJ, 2009). The reduced form hydrological model is a node-link network that controls the flows of water in each node and estimates the distribution of available surface water among users in each climate condition, calibrating it to observed water allocations in both normal and drought years. This approach to model river basin hydrology has been used in several studies such as Cai et al. (2003); and Ward and Pulido-Velazquez (2009).

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

$$WO_d = WI_d \cdot (1 - \gamma_d) - D_d^{IR} - D_d^{URB} \quad (3.13)$$

$$WI_{d+1} = WO_d + r_d^{IR} \cdot (D_d^{IR}) + r_d^{URB} \cdot (D_d^{URB}) \quad (3.14)$$

$$WO_d \geq E_d^{min} \quad (3.15)$$

The mass balance equation (3.13) determines the volume of water outflow WO_d from a river reach *d*, which is equal to the net (of evaporation loss γ_d) water inflow $WI_d \cdot (1 - \gamma_d)$ to *d* minus diversion for irrigation D_d^{IR} and for urban and industrial uses D_d^{URB}. The continuity equation (3.14) guarantees the continuity of river flow in the basin, where the volume of water inflow to the next river reach WI_{d+1} is the sum of outflow from the previous river reach WO_d, the return flows from previous irrigation districts $r_d^{IR} \cdot (D_d^{IR})$ and, the return flows from the cities $r_d^{URB} \cdot (D_d^{URB})$. Equation (3.15) states that the volume of water outflow WO_d from a river reach *d* must be greater than or equal to the minimum environmental flow E_d^{min} established for that river reach, which is determined by the basin's regulations.

We incorporate the reduced form hydrological model into a regional economic optimization model. For irrigation activities, a farm-level optimization model has been developed for each irrigation district. Irrigation districts maximize farmers' private benefits, subject to technical and resource constraints. The optimization problem for each irrigation district takes the following form:

$$Max\ B_k^{IR} = \sum_{ij} C'_{ijk} \cdot X_{ijk} \quad (3.16)$$

subject to

$$\sum_{ij} A_{ijk} \cdot X_{ijk} \leq R_k \quad (3.17)$$

$$X_{ijk} \geq 0 \quad (3.18)$$

where B_k^{IR} is farmers' private benefits in irrigation district *k*. C'_{ijk} is a vector of coefficients of net income per hectare of crop *i* cultivated under irrigation technology *j*. A_{ijk} is a matrix of production coefficients and R_k is a vector of constraint levels including land, water and labor in each irrigation district *k*. X_{ijk} corresponds to the area of crop *i* cultivated under irrigation technology *j* in irrigation district *k* and it is the decision variable in the irrigation district optimization problem.

For urban water uses, an economic surplus model has been developed for each city. The model maximizes the social (consumer and producer) surplus from water use for each city, subject to several physical and institutional constraints. The optimization problem for each urban center takes the following form:

$$Max\ B_u^{URB} = \left(a_{du} \cdot Q_{du} - \frac{1}{2} \cdot b_{du} \cdot Q_{du}^2 - a_{su} \cdot Q_{su} - \frac{1}{2} \cdot b_{su} \cdot Q_{su}^2\right) \quad (3.19)$$

subject to

$$Q_{du} - Q_{su} \leq 0 \quad (3.20)$$

$$Q_{du}\ ;\ Q_{su} \geq 0 \quad (3.21)$$

where B_u^{URB} is the social surplus of city u from water use. Q_{du} and Q_{su} are the quantity of water demanded and supplied by/to the city u, respectively. a_{du} and b_{du} are the intercept and the slope of the inverse demand function of city u, respectively. a_{su} and b_{su} are the intercept and the slope of the water supply function for city u, respectively. Equation (3.20) states that the quantity of water supplied must be greater than or equal to the quantity demanded.

The river basin optimization model accounts also for the environmental benefits provided by the main aquatic ecosystem in the JRB, the Albufera wetland. The model 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:

$$E_{Albufera} = \alpha \cdot r_{ARJ}^{IR} \cdot \left(D_{ARJ}^{IR}\right) + \beta \cdot r_{RB}^{IR} \cdot \left(D_{RB}^{IR}\right) \quad (3.22)$$

$$B_{Albufera} = \begin{cases} \delta_1 & if\ 0 \leq E_{Albufera} \leq E_1 \\ \delta_2 + \rho_2 \cdot E_{Albufera} & if\ \ E_1 < E_{Albufera} \leq E_2 \\ \delta_3 + \rho_3 \cdot E_{Albufera} & if\ \ E_2 < E_{Albufera} \leq E_3 \end{cases} \quad (3.23)$$

where equation (3.22) determines the quantity of water flowing to the Albufera wetland, $E_{Albufera}$. Parameters α and β represent the shares of return flows that feed the wetland from the ARJ and RB irrigation districts, respectively. The products $r_{ARJ}^{IR} \cdot (D_{ARJ}^{IR})$ and $r_{RB}^{IR} \cdot (D_{RB}^{IR})$ are return flows from the ARJ and RB irrigation districts, respectively. Equation (3.23) represents environmental benefits, $B_{Albufera}$, that the Albufera wetland provides to society. The environmental benefit function is assumed to be a piecewise linear function of the water inflows, $E_{Albufera}$, to the wetland. This function expresses shifts in the ecosystem status when critical thresholds of environmental conditions are reached (water inflows E_1, E_2 and E_3). 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. This function has been built following the methodology developed by Jorgensen et al. (2010) using time series data of various ecosystem health indicators of the wetland from the JRB authority reports, and economic valuation estimates of wetland services from the literature. Figure A1 in the appendix shows the environmental benefit function of the Albufera wetland.

The river basin optimization 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:

$$Max\left(\sum_k B_k^{IR} + \sum_u B_u^{URB} + B_{Albufera}\right) \tag{3.24}$$

subject to the constraints in equations (3.13), (3.14), (3.15), (3.17), (3.18), (3.20), (3.21) and (3.22).

The river basin optimization model allows calculating basin benefits under current institutional setting or baseline scenario (the non-cooperative solution) and it is the basis for calculation of benefits accrued to users under various cooperative arrangements for different drought scenarios.

Detailed biophysical and economic data has been collected from several sources including water inflows and diversions, crop area and water requirements, irrigation efficiency, crop costs and revenues, and water costs and prices by sector. Selected hydrologic and economic parameters of the JRB model are shown in Table 3.1. The river basin model and the CGT application have been run using the GAMS package.

3.3.3 Scenario simulation

The main water users in the JRB (described in section 3.1) are classified into four players that have similar characteristics regarding water use and their relation with the Albufera wetland. Players in the JRB game are: irrigation districts linked to the Albufera including the ARJ and RB irrigation districts (IE); irrigation districts not linked to the Albufera including the EM, CJT, and ESC irrigation districts (INE); the cities including Valencia, Sagunto and Albacete (C); and the Albufera wetland (E). This classification will allow us to capture all important strategic relationship between players in various locations of the basin and their opposed interests, and at the same

Table 3.1. 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
Share of return flows feeding the Albufera		
ARJ (α)	28	%
RB (β)	23	%
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

time to keep the computational burden at a reasonable level.

The cooperative water sharing agreement described in section 2 (equations (3.1) and (3.2)) is applied for two different scenarios of water management. The purpose is to find efficient and fair allocations of water and income among the players, and to explore the likelihood for ecosystem protection success. The scenarios are the following:

Scenario 1**:** This scenario maximizes the private benefits of the basin under all possible coalitional arrangements. The private benefits are the sum of the benefits of players IE, INE and C, disregarding the environmental benefits provided to society by player E (the Albufera wetland). The wetland receives water from return flows generated by player IE, similar to what happens in the current situation. The wetland is a weak player in the game because it does not compete for water.

Scenario 2**:** This scenario maximizes the social benefits of the basin under all possible coalitional arrangements. The social benefits are the sum of the benefits of all the players in the game, including the environmental benefits provided to society by player E (the Albufera wetland). In this case, the wetland is competing for water with other

Table 3.2. Benefits under the baseline situation for different climate conditions (10^6 €).

Users	Normal flow	Mild drought	Severe drought	Very severe drought
EM	79.8	71.9	66.4	60.7
CJT	44.9	40.6	37.2	35.7
ARJ	34.1	31.0	27.0	22.9
ESC	7.3	6.8	5.7	4.2
RB	24.2	20.7	16.5	12.1
Irrigation sector	**190.3**	**170.9**	**152.8**	**135.6**
Valencia	216.3	214.0	206.6	186.9
Sagunto	26.1	24.1	22.2	16.8
Albacete	40.2	38.9	38.8	38.6
Urban sector	**282.6**	**277.0**	**267.6**	**242.3**
Albufera wetland	**74.7**	**37.2**	**33.0**	**33.0**
Total JRB	**547.7**	**485.1**	**453.4**	**410.9**

users, and does not depend passively on remaining return flows.

These two scenarios are simulated under normal flow and various drought conditions using two sets of coalitional arrangements. Drought is classified into three levels, depending on the severity of the drought event: mild, severe, and very severe, based on historical data about water inflows in the JRB. The two sets of coalitional arrangements are: (a) partial cooperation in which the two scenarios are run with different combination of players; and (b) full cooperation, in which the two scenarios are run with all the players.

3.4 Results and discussion

The baseline situation (non-cooperation) represents the current conditions of water allocations in the JRB. Each player is maximizing its private benefits from its administrative water allocation, and there is no cooperation in the form of water sharing among the players. The results of the baseline situation are presented in Tables 3.2 and 3.3. Benefits in the JRB under the baseline situation for normal flow conditions amount to 548 million € from using 1,149 Mm^3. Irrigation activities generate 190 million € from using 1,030 Mm^3. The social surplus of the cities is 283 million € and they use 119 Mm^3. Environmental benefits provided by the Albufera wetland are 75 million €. The Albufera wetland receives 60 Mm^3 from the return flows of the ARJ and RB irrigation districts, which support the good ecological status of the wetland.

Table 3.3. Water use under the baseline situation for different climate conditions (Mm^3).

Users	Normal flow	Mild drought	Severe drought	Very severe drought
EM	399	359	332	304
CJT	155	132	115	107
ARJ	200	180	155	130
ESC	33	30	25	18
RB	243	207	167	123
Irrigation sector	**1,030**	**908**	**794**	**682**
Valencia	94	81	67	53
Sagunto	8	7	6	4
Albacete	17	17	17	17
Urban sector	**119**	**105**	**90**	**74**
Albufera wetland	**60**	**52**	**43**	**34**
Total JRB	**1,149**	**1,013**	**884**	**756**

Note: Total water use in the JRB is the sum of water use in the irrigation and urban sectors, and does not include water return flowing to the Albufera wetland.

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. The full demand of Valencia (94 Mm^3) and Sagunto (8 Mm^3) is always covered.

Results of the drought scenarios indicate that drought events reduce the benefits of the JRB between 11 and 25%. Water use patterns show a reduction in extractions between 12 and 34%. Irrigation activities reduce water extractions between 12 and 34%. Irrigation benefit losses range between 10 to 30% of benefits in normal year. The reduction in irrigation water extractions has large negative impacts on the Albufera wetland that is mostly fed by irrigation return flows. Water inflows to the Albufera wetland decrease between 13 and 43%, depending on drought severity. As a consequence, drought damages for the Albufera wetland under drought conditions exceed 50% of benefits in normal years.

The current water resources regulation in the JRB guarantees the availability of urban water to human population. During severe drought spells, the urban demand must be first fully covered because of such priority rules. The three simulated drought scenarios show a reduced supply from the Jucar River to the main cities in the JRB. However, the full demand of Valencia and Sagunto is always covered with additional water from the neighboring Turia River Basin. During extreme drought periods, the provision of water to these cities is shared equally between the Jucar and the Turia Rivers. In the city of Albacete, the supply of water during dry periods is amended by

Table 3.4. Results of the characteristic functions under non-cooperation and full cooperation for the scenarios of water management (10^6 €).

Water management scenarios	**Coalitional arrangements**		**Normal**	**Mild drought**	**Severe drought**	**Very severe drought**
Scenario 1	Non-cooperation	{INE}	132.0	119.2	109.3	100.5
		{IE}	58.3	51.7	43.5	35.0
		{C}	282.6	277.0	267.6	242.3
		{E}	74.7	37.2	33.0	33.0
		Total	**547.7**	**485.1**	**453.4**	**410,9**
	Full cooperation	**{INE,IE,C,E}**	**582.4** (6%)	**517.8** (7%)	**474.5** (5%)	**427.3** (4%)
Scenario 2	Non-cooperation	{INE}	132.0	119.2	109.3	100.5
		{IE}	58.3	51.7	43.5	35.0
		{C}	282.6	277.0	267.6	242.3
		{E}	74.7	37.2	33.0	33.0
		Total	**547.7**	**485.1**	**453.4**	**410.9**
	Full cooperation	**{INE,IE,C,E}**	**742.3** (36%)	**735.0** (52%)	**710.1** (57%)	**659.6** (61%)

Note: The percentage gain in benefits between full cooperation and non-cooperation is given in parenthesis.

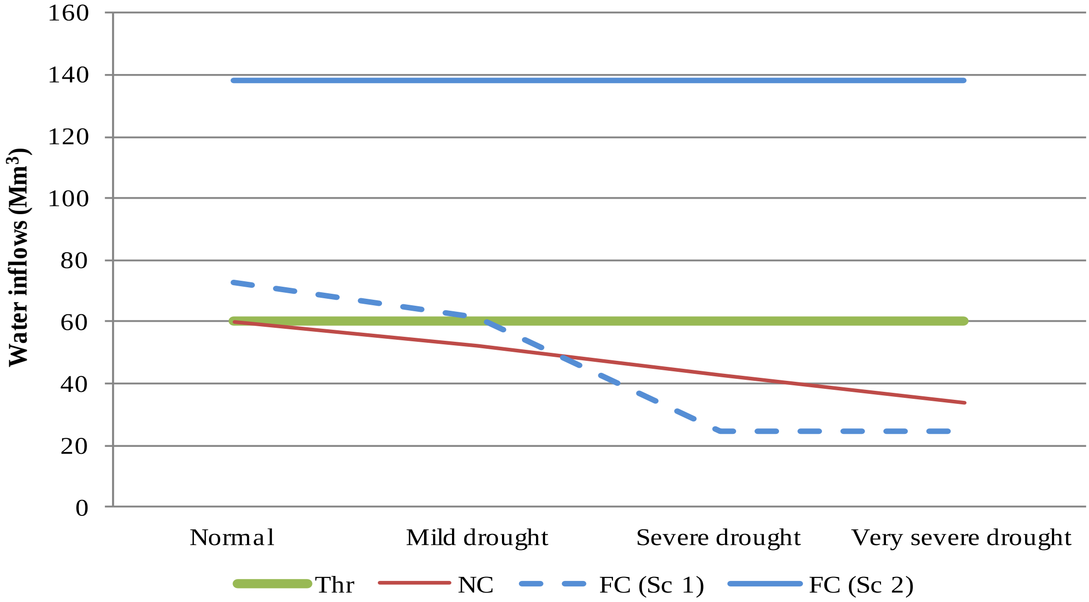
pumping groundwater from the Eastern La Mancha aquifer (CHJ, 2009). The simulation results for the urban sector indicate that the provision of surface water from the Jucar River falls between 14 and 45%, while groundwater extractions increase up to 8 Mm^3. The benefit losses during droughts in the urban sector are below 14% 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.

3.4.1 Cooperative water management

Table 3.4 presents the values of the characteristic function under non-cooperation (baseline) and full cooperation for different drought conditions in the two scenarios of water management. Detailed results of the characteristic function of all coalitional arrangements under drought conditions for the two scenarios are presented in Tables A1 and A2 in the appendix.

The results suggest that full cooperative management of water in the JRB achieves the highest aggregate level of benefits for the two scenarios and all drought conditions. For *Scenario 1*, full cooperation among users improves benefits between 16 and 34 million € (4 to 7%) compared to non-cooperation. When a policy to protect the Albufera wetland is introduced in *Scenario 2*, full cooperation improves significantly benefits between 195 and 285 million € (36 to 61%) compared to non-cooperation.

Figure 3.1. Water inflows to the Albufera wetland under different coalitional arrangements and drought conditions for scenarios 1 and 2.



Note: Thr= Threshold, NC= Non-cooperation, FC (Sc 1)= Full cooperation in *Scenario 1*, FC (Sc 2)= Full cooperation in *Scenario 2*. The threshold considered is 60 Mm^3 and it is calculated based on the minimum water requirements of the Albufera wetland and the percentage contribution of irrigation activities to water flowing to the wetland.

These improvements in benefits of full cooperation under both scenarios occur mainly because player IE transfers part of its water to players INE and E. Benefits under partial cooperation are always higher than under non-cooperation, but lower than under full cooperation.

The values of the characteristic functions of the JRB game under the different cooperative arrangements for the water management and climate scenarios show superadditivity compared to non-cooperation. This property is important because it indicates that the players have an incentive to cooperate. This incentive increases considerably when the environmental benefits provided by the Albufera wetland to society are accounted for in *Scenario 2*. Furthermore, it seems that partial cooperation between players IE, INE, and E is sufficient to maximize the benefits of the JRB and protect the Albufera wetland, and player C could be excluded from the game due to its minute contribution.[3] However, these results do not guarantee the acceptability of the cooperative agreement by the players nor its stability, and the likelihood of failure of cooperation remains. Therefore, to assure that the players remain cooperative, the

[3] Player C is called a dummy player, using the Game Theory Jargon.

reallocation of benefits among the players should be performed using the CGT solution concepts. These allocations are calculated in section 4.2.

Figure 3.1 presents the quantity of water flowing to the Albufera wetland under different cooperative arrangements and drought conditions for scenarios 1 and 2. Results indicate clearly that policy intervention to protect the Albufera wetland (*Scenario 2*) is better than non-intervention, securing always a fixed amount of water (138 Mm^3) flowing to the wetland. This amount is well above the minimum technical requirement of the Albufera wetland (60 Mm^3) set by the basin authority, and thus ensures a good ecological status. Moreover, cooperation without public intervention fails to provide the wetland with a minimum water threshold that could maintain its good ecological status (*Scenario 1*). Water inflows to the Albufera wetland in *Scenario 1* for severe and very severe droughts are far below the minimum requirement.

We find that achieving cooperation without policy intervention to regulate the Albufera wetland degrades the wetland. The reason is that most services provided by the Albufera wetland are public goods, and the private decision-makers in the river game have little incentive to conserve water and enhance the provision of such services. The Albufera wetland is linked to the IE player (ARJ and RB) which displays a lower value of water than the INE player (EM, CJT, and ESC). This is a common situation for environmental assets worldwide which are usually linked to subsidiary or low-value activities. In *Scenario 1*, benefit gains are achieved by reallocating water from player IE to player INE. Consequently, return flows to the wetland decline as drought severity intensifies producing the desiccation and degradation of ecosystems. Hence, both policy intervention and cooperation (*Scenario 2*) are needed for the full protection of the Albufera wetland under drought.

The comparison between the two scenarios indicates that public intervention to protect the Albufera through its inclusion in the cooperative agreement (*Scenario 2*), provides high incentives for cooperation. The result is a more sustainable use of water and substantial gains in basin benefits. A major policy implication from the analysis is that cooperation may have to be regulated by public agencies (the basin authority in this case) when scarcity is very high, in order to protect ecosystems and increase regional economic benefits.

Table 3.5. Benefits by CGT solutions and non-cooperation in *Scenario 1*.

Climate scenarios	Players	Non-cooperation	Full cooperation		
			Shapley	Nash-Harsanyi	Nucleolus
Normal	INE	132.0	143.5	140.7	132.2
	IE	58.3	70.0	67.0	58.5
	C	282.6	282.7	291.3	282.6
	E	74.7	86.3	83.4	109.1
Mild drought	INE	119.2	130.8	127.4	121.4
	IE	51.7	64.5	59.9	82.2
	C	277.0	277.3	285.2	277.0
	E	37.2	45.2	45.4	37.2
Severe drought	INE	109.3	118.7	114.6	127.3
	IE	43.5	53.1	48.8	43.6
	C	267.6	269.5	272.9	270.6
	E	33.0	33.2	38.3	33.0
Very severe drought	INE	100.5	107.1	104.6	112.1
	IE	35.0	43.2	39.1	38.1
	C	242.3	243.8	246.4	244.1
	E	33.0	33.1	37.1	33.0

3.4.2 Allocations of the cooperative benefits

The results of the different cooperative arrangements suggest that cooperative water management in the JRB yields higher benefits compared to non-cooperation. The challenge here is to allocate the benefits from cooperation among the players in a fair manner. The allocation of benefits is calculated using the different CGT allocation solutions. Then, the acceptability and stability of the benefit allocations are tested using the Core conditions (equations (3.3) to (3.5)), the power index (θ_l^a), and the stability index ($\overline{\theta_a}$). Tables 3.5 and 3.6 show the allocated benefits to each player, based on the different CGT solutions.

Results of benefit allocations highlight that the preferred CGT solutions for the players vary, depending on the scenario of water management and the drought condition. The reason for these results lies in the properties of the CGT solutions. Player C does not contribute to any coalition in all management and climate scenarios but gains an equal share of benefit with Nash-Harsanyi. This is because Nash-Harsanyi allocates an equal incremental gain to each player based on its original benefit under non-cooperation, irrespective of its contribution to the coalition. Player E does not contribute either under *Scenario 1*, but gets an equal share with Nash-Harsanyi. Player E prefers mostly Shapley under *Scenario 2*, because it makes a contribution that is

Table 3.6. Benefits by CGT solutions and non-cooperation in *Scenario 2*.

Climate scenarios	Players	Non-cooperation	Full cooperation		
			Shapley	Nash-Harsanyi	Nucleolus
Normal	INE	132.0	216.1	180.7	132.2
	IE	58.3	67.8	107.0	58.5
	C	282.6	282.8	331.3	282.6
	E	74.7	175.7	123.4	269.0
Mild drought	INE	119.2	209.6	181.7	291.4
	IE	51.7	84.1	114.2	93.9
	C	277.0	283.8	339.5	281.3
	E	37.2	157.5	99.7	68.6
Severe drought	INE	109.3	185.6	173.5	231.9
	IE	43.5	95.0	107.7	88.2
	C	267.6	303.9	331.8	312.3
	E	33.0	125.6	97.2	77.7
Very severe drought	INE	100.5	155.8	162.7	162.7
	IE	35.0	113.5	97.2	97.2
	C	242.3	283.8	304.5	304.5
	E	33.0	106.5	95.2	95.2

rewarded in the Shapley solution. Player INE prefers mostly the Nucleolus because this solution discourages the formation of partial coalitions that do not benefit him. These empirical findings on the preferred cooperative solutions for the players indicate the different interests of the players, and the difficulties to achieve a sustainable cooperative agreement at basin scale in the Jucar basin.

The analysis of the acceptability of the CGT allocations using the Core requirements indicates that the benefit allocations based on the Shapley and Nash-Harsanyi solutions for *Scenario 1* under different drought conditions satisfy only individual rationality (equation 3.3) and the efficiency condition (equation 3.5), but not group rationality (equation 3.4). These allocations are not in the Core of the game, and they are not acceptable by the players. Therefore, the Shapley and Nash-Harsanyi solutions are not stable, and players may consider defection from the grand coalition to create partial coalitions. However, the Core requirements are satisfied for benefit allocations based on the Nucleolus solution, and they are acceptable to players in *Scenario 1*. For these reasons, the most stable cooperative solution in *Scenario 1* is the Nucleolus for all drought scenarios.

Under *Scenario 2*, the benefit allocations based on the three cooperative solutions satisfy the Core requirements, and since these allocations are in the Core they are acceptable to all players. So, theoretically there are no incentives for the players to leave

Table 3.7. Power and stability indexes in *Scenario 2*.

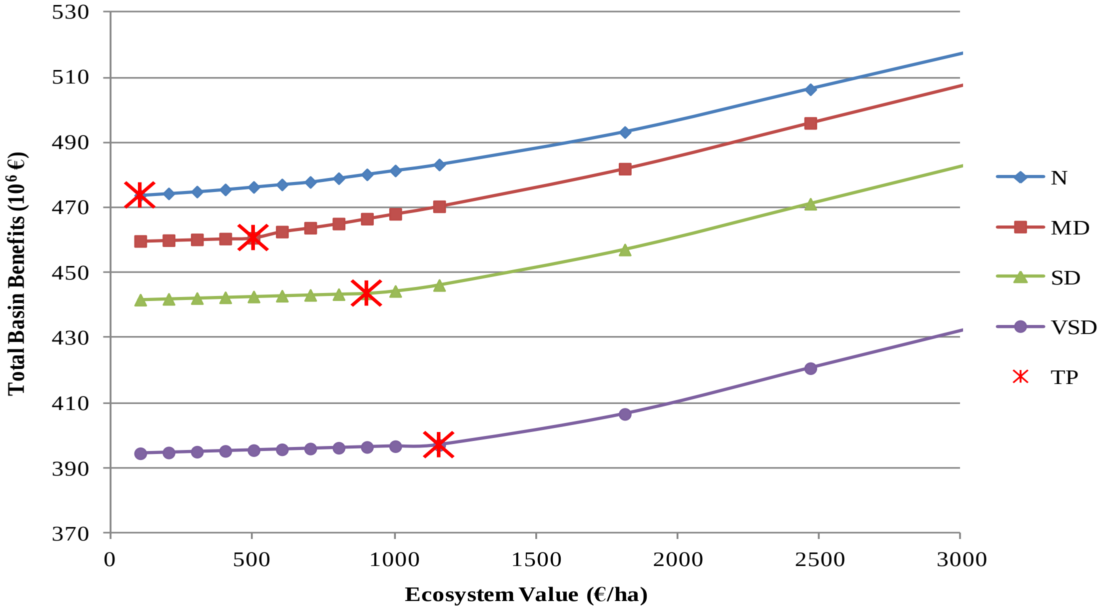
Cooperative solution	**Power indexes of players (θ_i^a)**				**Stability index $\overline{\theta_a}$**
	INE	IE	C	E	
Normal Flow					
Shapley	0.43	0.05	0.00	0.52	1.05
Nash-Harsanyi	0.25	0.25	0.25	0.25	0.00
Nucleolus	0.00	0.00	0.00	1.00	1.99
Mild drought					
Shapley	0.36	0.13	0.03	0.48	0.83
Nash-Harsanyi	0.25	0.25	0.25	0.25	0.00
Nucleolus	0.69	0.17	0.02	0.13	1.20
Severe drought					
Shapley	0.30	0.20	0.14	0.36	0.39
Nash-Harsanyi	0.25	0.25	0.25	0.25	0.00
Nucleolus	0.48	0.17	0.17	0.17	0.61
Very severe drought					
Shapley	0.22	0.32	0.17	0.30	0.27
Nash-Harsanyi	0.25	0.25	0.25	0.25	0.00
Nucleolus	0.25	0.25	0.25	0.25	0.00

the grand coalition in order to act individually or to participate in partial coalitions. However, players have different preferences over the various allocation solutions. Therefore, there is a need to evaluate the stability of these solutions to find the best one in this scenario. Table 3.7 presents the power and the stability indexes for each cooperative solution in *Scenario 2*.

The stability indexes show that the most stable cooperative solution is the Nash-Harsanyi for all drought scenarios, although for a very severe drought scenario the Nucleolus achieves the same degree of stability as the Nash-Harsanyi. The least stable cooperative solution is the Nucleolus under normal flow, and mild and severe droughts, and the Shapley is the least stable under very severe drought conditions. Scrutiny of the stability indexes indicates that the stability of the grand coalition increases as drought severity intensifies. This means that the severity of drought is an incentive to act cooperatively.

The power indexes of players under the Shapley solution indicate that player E (the Albufera wetland) has the highest propensity to cooperate and stay in the grand coalition under all drought conditions, while player C (the cities) has the lowest propensity to cooperate and may disrupt the grand coalition unless improving its allocation. Under the Nash-Harsanyi solution, the power is distributed equally among

Figure 3.2. Sensitivity analysis of the ecosystem value of the Albufera wetland.



Note: N=Normal flow year, MD=Mild drought, SD=Severe drought, VSD=Very severe drought. TP=Tipping point.

the players, which means that the grand coalition is more likely to be stable. The Nucleolus solution shows that players E, IE, and INE display a high propensity to cooperate.

The results of the analysis of the acceptability and stability of the cooperative solutions suggest that the internalization of environmental damages in *Scenario 2* provides more stability to cooperation compared to *Scenario 1*. However, stability of cooperation under *Scenario 2* would likely be affected by the economic value of the ecosystem. A sensitivity analysis has been conducted in order to assess the results under *Scenario 2*, and their robustness to different economic valuation estimates of the Albufera wetland (Figure 3.2). Results indicate that ecosystem value and drought condition affect the policy decision concerning the protection of the wetland. The tipping points in figure 3.2 show critical ecosystem values below which the Albufera wetland is excluded from the water sharing agreement, and the game stability is reduced. The tipping point moves to higher values of the Albufera as drought severity intensifies because of the increase in the economic value of water (shadow price) to users.

3.5 Conclusions and policy implications

This paper develops a cooperative game theory framework in order to analyze the possibilities of cooperation over sharing water resources, and the options for protecting ecosystems in arid and semiarid basins under scarcity and drought. The framework was empirically tested in the Jucar River Basin (Spain), a typical highly stressed river basin in a semiarid region with acute water scarcity problems that are damaging valuable ecosystems.

Results indicate that drought damage costs in the Jucar River Basin are considerable. However, the cooperation of stakeholders through the right institutional setting reduces drought damage costs between 4 and 7%. When environmental damages are internalized through the inclusion of the wetland in the cooperative agreement, the cooperative results are more appealing, reducing drought damage costs by 52 to 61%.

Cooperative water management may be challenging in practice because of the strategic behavior of stakeholders and the high transaction costs of organizing collective action. Water agencies can promote cooperative management by creating different incentives for cooperation, such as taxes and subsidies, diversion thresholds, monitoring mechanisms, and technical advice. The role of these agencies is especially important in protecting ecosystems. Our empirical results indicate that cooperative management improves the economic benefits of water users but it may have little effect on ecosystems protection without additional incentives or regulations.

The cooperative game theory solutions and stability indexes examined in this paper provide information about the possibility for cooperation in the Jucar River Basin. This information could be helpful to reach an agreement to share water resources that could enhance private and social benefits. The empirical results suggest that cooperation is a feasible option, but the basis for cooperation is weak hindering the acceptability and stability of the cooperative agreement. However, the internalization of environmental damages provides more stability to the agreement, although it depends on the value of ecosystem.

The results highlight the fact that various cooperative solutions have different outcomes in terms of their acceptability by the players and their stability. This finding has important policy implication because it demonstrates the difficulties in selecting a mix of policy instruments that could address scarcity, and mitigate the negative impacts of droughts, and the risk of policy failure.

While the empirical analysis was performed using the Jucar Basin situation, our analytical framework is capable of providing meaningful results to any of the mounting cases of climate change-related water scarcity issues in any of the basins in arid and semiarid regions, including the ones mentioned in this paper. The inclusion of the strategic behavior of the parties involved in the drought mitigation policies is new to the policy analysis and would add an important aspect to the analysis of policy feasibility under scarce water situations.

3.6 References

Alcamo, J., Henrichs, T., Rösch, T., 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.

Ambec, S., Ehlers, L., 2008. Sharing a river among satiable agents. *Games and Economic Behavior* 64 (1), 35-50.

Booker, J., Michelsen, A., Ward F., 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

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

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

Dinar, A., Farolfi, S., Patrone, F., Rowntree, K., 2008. To negotiate or to game theorize: evaluating water allocation mechanisms in the Kat basin, South Africa. In Dinar, A., Albiac, J., Sanchez-Soriano, J. (Eds.), Game theory and policymaking in natural resources and the environment, Routledge, New York, pp. 85-111.

Dinar, A., Howitt, R., 1997. Mechanisms for allocation of environmental control cost: empirical tests of acceptability and stability. *J. Environ. Manage.* 49, 183-203.

Esteban, E., Dinar, A., 2013. Cooperative management of groundwater resources in the presence of environmental externalities. *Environ. Resour. Econ.* 43 (3), 443-469.

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

Finlayson, C., Davidson, N., 1999. Global review of wetlands resources and priorities for wetland inventory, Summary report, Wetlands International, Wageningen.

García-Molla, M., Sanchis, C., Ortega, M., Avella, L., 2013. Irrigation associations coping with drought: The case of four irrigation districts in Eastern Spain. In Schwabe, K., Albiac, J., Connor, J., Hassan, R., Meza, L. (Eds), Drought in Arid and

Semi-Arid Environments: A Multi-Disciplinary and Cross-Country Perspective, Springer, Dordrecht, pp. 101-122.

Gillies, D., 1958. Solutions to general non-zero-sum games. In Tucker, A., Luce, R., (Eds.), Contributions to the Theory of Games, Princeton University Press, New Jersey, pp. 47-85.

Harsanyi, J., 1958. A bargaining model for the cooperative n-person game. In Tucker, A., Luce, R., (Eds), Contributions to the Theory of Games, Princeton University Press, New Jersey, pp. 324-356.

Howitt, R., Medellin, J., Lund, J., MacEwan D., 2014. Preliminary 2014 drought economic impact estimates in Central Valley agriculture, Report prepared for the California Department of Food and Agriculture, Center for Watershed Sciences, University of California, Davis.

Intergovernmental Panel on Climate Change (IPCC), 2014. Climate Change 2014: impacts, adaptation, and vulnerability, Contribution of working groups II to the fifth assessment report of the IPCC, IPCC, Geneva.

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

Kirby, M., Bark, R., Connor, J., Qureshi, E., Keyworth, S., 2014. Sustainable irrigation: How did irrigated agriculture in Australia's Murray–Darling Basin adapt in the Millennium Drought?. *Agr. Water Manage* 145, 154-162.

Loehman, E., Orlando, J., Tschirhart, J., Winstion, A., 1979. Cost allocation for a regional wastewater treatment system. *Water Resour. Res.* 15, 193-202.

Madani, K., 2010. Game theory and water resources. *J. Hydrol.* 381, 225-238.

Madani, K., Hooshyar, M., 2014. A game theory–reinforcement learning (GT–RL) method to develop optimal operation policies for multi-operator reservoir systems. *J. Hydrol.* 519, 732-742.

Nash, J., 1953. Two-person cooperative games. *Econometrica* 21(1), 128-140.

Parrachino, I., Dinar, A., Patrone, F., 2006. Cooperative game theory and its application to natural, environmental and water resource issues: application to water resources, World Bank Policy Research Working Paper 4074, World Bank, Washington.

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

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

Schmeidler, D., 1969. The nucleolus of a characteristic function game. *SIAM J. Appl. Math.* 17: 1163-1170.

Schwabe, K., Albiac, J., Connor, J., Hassan, R., Meza, L., (Eds), 2013. Drought in arid and semi-arid regions: A multi-disciplinary and cross-country perspective, Springer, Dordrecht.

Shapley, L., 1973. Cores of convex games. *Int. J. Game Theory* 1 (1), 11-26.

Shapley, L., A value for n-person games. *Annals of Mathematics Studies* 28, 307-318.

Shapley, L., Shubik, M., 1954. A method for evaluating the distribution of power in a committee system. *Am. Polit. Sci. Rev.* 48, 787-792.

Straffin, P., Heaney, J., 1981. Game theory and the Tennessee valley authority. *Int. J. Game Theory* 10 (1), 35-43.

United Nations Development Programme (UNDP), 2006. Human development report 2006: beyond scarcity: power, poverty and the global water crisis, UNDP, New York.

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

Wang, L., Fang, L., Hipel, K., 2008. Basin-wide cooperative water resources allocation. *Eur. J. Oper. Res.* 190, 798-817.

Ward, F., Pulido-Velazquez, M., 2009. Incentive pricing and cost recovery at the basin scale. *J. Environ. Manage.* 90, 293-313.

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

Zilberman, D., Dinar, A., MacDougall, N., Khanna, M., Brown, C., Castillo, F., 1998. Private and institutional adaptation to water scarcity during the California drought, 1987-92, Staff Paper 9802, Resource Economics Division, Economic Research Service, USDA, Washington.

3.7 Appendix

Figure A1. Environmental benefits function of the Albufera wetland.

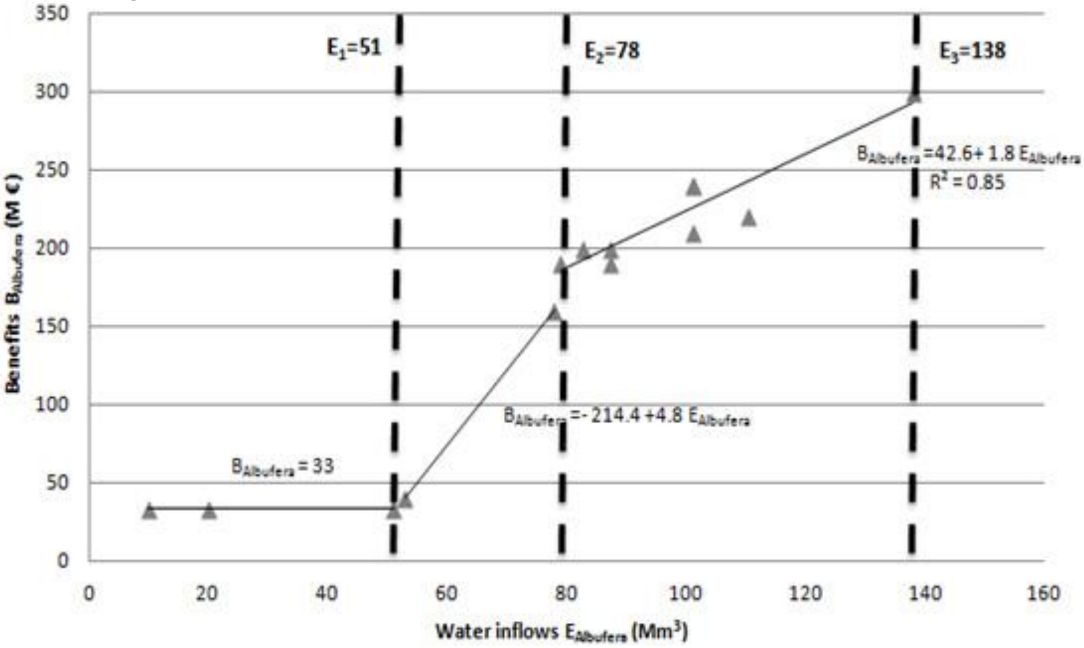


Table A1. Results of the characteristic functions under different coalitional arrangements and drought conditions in *Scenario 1* (10^6 €).

Coalitional arrangements	Players	Normal flow	Mild drought	Severe drought	Very severe drought
Non-cooperation	{INE}	132.0	119.2	109.3	100.5
	{IE}	58.3	51.7	43.5	35.0
	{C}	282.6	277.0	267.6	242.3
	{E}	74.7	37.2	33.0	33.0
	Total	**547.7**	**485.1**	**453.4**	**410.9**
Partial cooperation	{INE,IE}	190.6	181.9	170.3	150.2
	{C}	282.7	277.0	267.6	242.3
	{E}	74.5	33.0	33.0	33.0
	Total	**547.8**	**491.9**	**470.9**	**425.5**
Partial cooperation	{INE,C}	414.8	398.4	379.0	344.1
	{IE}	58.3	51.7	43.5	35.0
	{E}	74.7	37.2	33.0	33.0
	Total	**547.8**	**487.3**	**455.5**	**412.1**
Partial cooperation	{INE,E}	206.8	158.6	144.4	134.8
	{IE}	58.3	51.7	43.5	35.0
	{C}	282.6	277.0	267.6	242.3
	Total	**547.7**	**487.3**	**455.5**	**412.1**
Partial cooperation	{IE,C}	341.1	330.0	314.2	282.2
	{INE}	149.1	119.2	109.3	100.5
	{E}	74.8	40.8	33.0	33.0
	Total	**565.0**	**490.0**	**456.5**	**415.7**
Partial cooperation	{IE,E}	133.5	94.0	76.6	68.1
	{C}	282.6	277.0	267.6	242.3
	{INE}	132.0	119.2	109.3	100.5
	Total	**548.1**	**490.2**	**453.5**	**410.9**
Partial cooperation	{C,E}	357.4	314.2	300.6	275.3
	{INE}	132.0	119.2	109.3	100.5
	{IE}	58.3	51.7	43.5	35.0
	Total	**547.7**	**485.1**	**453.4**	**410.8**
Partial cooperation	{INE,IE,C}	473.3	459.5	441.5	394.3
	{E}	74.5	33.0	33.0	33.0
	Total	**547.8**	**492.5**	**474.5**	**427.3**
Partial cooperation	{INE,IE,E}	299.8	240.8	203.3	183.2
	{C}	282.6	277.0	267.6	242.3
	Total	**582.4**	**517.8**	**470.9**	**425.5**
Partial cooperation	{INE,C,E}	489.5	435.6	412.0	377.1
	{IE}	58.3	51.7	43.5	35.0
	Total	**547.8**	**487.3**	**455.5**	**412.1**
Partial cooperation	{E,C,IE}	416.1	370.9	347.2	315.2
	{INE}	132.0	119.2	109.3	100.5
	Total	**548.1**	**490.1**	**456.5**	**415.7**
Full cooperation	**{INE,IE,C,E}**	**582.4** (6%)	**517.8** (7%)	**474.5** (5%)	**427.3** (4%)

Note: The percentage gain in benefits between full cooperation and non-cooperation is given in parenthesis.

Table A2. Results of the characteristic functions under different coalitional arrangements and drought conditions in *Scenario 2* (10^6 €).

Coalitional arrangements	Players	Normal	Mild drought	Severe drought	Very severe drought
Non-cooperation	{INE}	132.0	119.2	109.3	100.5
	{IE}	58.3	51.7	43.5	35.0
	{C}	282.6	277.0	267.6	242.3
	{E}	74.7	37.2	33.0	33.0
	Total	**547.7**	**485.1**	**453.4**	**410.9**
Partial cooperation	{INE,IE}	190.6	181.9	170.3	150.2
	{C}	282.7	277.0	267.6	242.3
	{E}	74.5	33.0	33.0	33.0
	Total	**547.8**	**491.9**	**470.9**	**425.5**
Partial cooperation	{INE,C}	414.8	398.4	379.0	344.1
	{IE}	58.3	51.7	43.5	35.0
	{E}	74.7	37.2	33.0	33.0
	Total	**547.8**	**487.3**	**455.5**	**412.1**
Partial cooperation	{INE,E}	389.6	312.3	190.0	134.8
	{IE}	58.3	51.7	43.5	35.0
	{C}	282.6	277.0	267.6	242.3
	Total	**730.5**	**641.0**	**501.1**	**412.1**
Partial cooperation	{IE,C}	341.1	330.0	314.2	282.2
	{INE}	132.0	119.2	109.3	100.5
	{E}	74.8	40.8	33.0	33.0
	Total	**547.9**	**490.0**	**456.5**	**415.7**
Partial cooperation	{IE,E}	166.7	157.5	79.1	68.1
	{C}	282.6	277.0	267.6	242.3
	{INE}	132.0	119.2	109.3	100.5
	Total	**581.3**	**553.7**	**456.0**	**410.9**
Partial cooperation	{C,E}	358.6	314.2	300.6	275.3
	{INE}	132.0	119.2	109.3	100.5
	{IE}	58.3	51.7	43.5	35.0
	Total	**548.9**	**485.1**	**453.4**	**410.8**
Partial cooperation	{INE,IE,C}	473.3	459.5	441.5	394.3
	{E}	74.5	33.0	33.0	33.0
	Total	**547.8**	**492.5**	**474.5**	**427.3**
Partial cooperation	{INE,IE,E}	459.7	449.5	353.1	283.4
	{C}	282.6	277.0	267.6	242.3
	Total	**742.3**	**726.5**	**620.7**	**525.7**
Partial cooperation	{INE,C,E}	672.3	636.9	540.7	386.5
	{IE}	58.3	51.7	43.5	35.0
	Total	**730.6**	**688.6**	**584.2**	**421.5**
Partial cooperation	{E,C,IE}	449.3	439.4	422.6	389.5
	{INE}	132.0	119.2	109.3	100.5
	Total	**581.3**	**558.6**	**531.9**	**490.0**
Full cooperation	**{INE,IE,C,E}**	**742.3** (36%)	**735.0** (52%)	**710.1** (57%)	**659.6** (61%)

Note: See note to table A1

.

Chapter 4

Efficient water management policies for irrigation adaptation to climate change in Southern Europe

Abstract

This paper evaluates economic and environmental effects of two incentive-based water management policies to address climate change impacts on irrigated agriculture: water markets and irrigation subsidies. A Southern European case study assesses farmers' long and short-run adaptation responses under climate change and policy interventions with a discrete stochastic programming model. Results indicate that climate change will likely have negative impacts on irrigation activities and water-dependent ecosystems in Southern Europe. However, the severity of impacts depends on government policy settings and farmers' adaptation responses. The comparison between water market and irrigation subsidy policies shows the advantages of water markets over irrigation subsidies in terms of both private and social benefits. These findings could contribute to the design of efficient climate change adaptation policies in the irrigated agriculture of Southern Europe.

Keywords: Climate change, Irrigation, Adaptation, Southern Europe, Stochastic programming, Water policies

4.1 Introduction

Climate change is a major challenge for sustainable agricultural production in the coming decades in arid and semiarid regions worldwide. In those regions, climate change will likely increase temperature and evapotranspiration, reduce precipitation and snowmelt, and modify precipitation patterns, impacting negatively on water resources, irrigated and dryland agriculture, and water-dependent ecosystems (IPCC, 2014). This challenge will be difficult to manage in a context of rising world food demand and growing competition between consumptive and environmental water uses (Elliot et al., 2014).

The South of Europe is one of the arid and semiarid regions where the vulnerability of irrigated agriculture to climate change is expected to be especially strong (IPCC, 2014). Climate change projections for this region suggest significant reductions in freshwater supplies from surface and groundwater resources, and increases of the frequency and longevity of extreme drought events (Lehner et al., 2006). The reductions of water availability and reliability in Southern Europe will be combined with increases of irrigation demand (Jimenez et al., 2014), leading mostly to reduced crop yields and shifts of some cultivation activities northward (EEA, 2012).

Irrigation adaptation to climate change in Southern Europe has become one of the main objectives of the European water and agricultural regulations, such as the Water Framework Directive and the 2014-2020 Rural Development policy (EC, 2009 and 2013). The evaluation of the effectiveness of existing adaptation policies and whether additional adaptation policies are needed is of particular interest for policymakers and stakeholders in the region. The response to these issues requires the development of studies that provide a better understanding of the economic and environmental impacts of climate change on irrigation, the adaptation policy alternatives, and the cost implications.

Many studies in the literature have addressed the issue of irrigated agriculture adaptation to the foreseeable climate change impacts. A wide variety of adaptation options has been proposed. Farm-level adaptation options such as improving irrigation scheduling, crop mix change, use of new crop varieties, and improving irrigation efficiency seem to contribute significantly to adaptation (Howden et al., 2007; Reidsma et al., 2010; Leclere et al., 2013). However, a string of the literature calls for a reconsideration of water institutions and policies used at present, and the implementation of incentive-based policies for more effective uptake of adaptation (Zilberman et al., 2002; Booker et al., 2005). Two popular incentive-based policies to address climate change irrigation adaptation which are widely considered in the literature are water markets and public subsidies for investments in efficient irrigation systems.

Water markets seem to be a good option to smooth the economic impacts of climate change (Calatrava and Garrido, 2005; Gomez-Limon and Martinez, 2006; Gohar and Ward, 2010). Estimations of water market benefits during the last drought in the Murray-Darling basin of Australia, which is at present the most active water market in the world, are close to 1 billion US dollars per year (Connor and Kaczan, 2013).[4] A challenge to water markets is the third party effects such as the environmental impacts. Water markets reduce streamflows because previously unused water allocations are traded, and also because gains in irrigation efficiency at parcel level reduce return flows to the environment (Howe et al., 1986; Qureshi et al., 2010). Another worrying effect is the large surge in groundwater extractions, as shown in the last drought in the Murray-Darling basin.[5] These environmental impacts reduce the benefits of trading and increase adaptation costs. For instance, water authorities in Australia are implementing very expensive public programs on infrastructure upgrading investments and environmental water buyback, in order to recover water for the environment in the Murray-Darling basin (Wheeler et al., 2014).

Public policies that provide subsidies for investments in efficient irrigation systems (irrigation modernization) are considered also important options for climate change adaptation (Cazcarro et al., 2011, Graveline et al., 2014; Varela et al., 2014). The reason is that modernization reduces land abandonment, facilitates the adoption of diversified and high-value cropping patterns, and improves crop yields, leading to an increase in the value of agricultural production (Perry et al., 2014). In addition, modernization supports rural development and improves water quality (Playan et al., 2013). However, contrary to widespread expectations, modernization increases water depletion through enhanced crop evapotranspiration and reduction of return flows. These flows contribute

[4] Potential water market benefits in California during drought have been also estimated at 1 billion US dollars per year (Medellin et al., 2013).

[5] Blewett (2012) indicates that extractions between 2002 and 2007 were seven times above the allowed limits placed on groundwater users.

to in-stream flows and groundwater replenishment that could be essential for downstream consumptive and environmental uses (Huffaker, 2008; Perry et al., 2014).

The above-mentioned studies analyze the advantages and limitations of water markets and irrigation subsidies in detail. However, there are no studies in the European context that provide a comparative analysis of the effectiveness of these two incentive-based policies for irrigation adaptation to climate change, and the extent to which farmers could realize potential adaptation opportunities. To address this gap in the literature, this paper presents a stochastic modeling framework to analyze the contribution of these two incentive-based policies to adaptation, and the economic and environmental tradeoffs between these policies.

The results obtained could guide policymakers on the design of efficient water institutions and policies to address climate change in the irrigated agriculture of Southern Europe. The lower Jucar basin in Spain is chosen as a representative basin for Southern Europe. This basin is a good experimental field for studying irrigation adaptation possibilities to the impending climate change. The Jucar River is under severe stress with acute water scarcity and near zero mouth outflows, and severe ecosystem degradation.

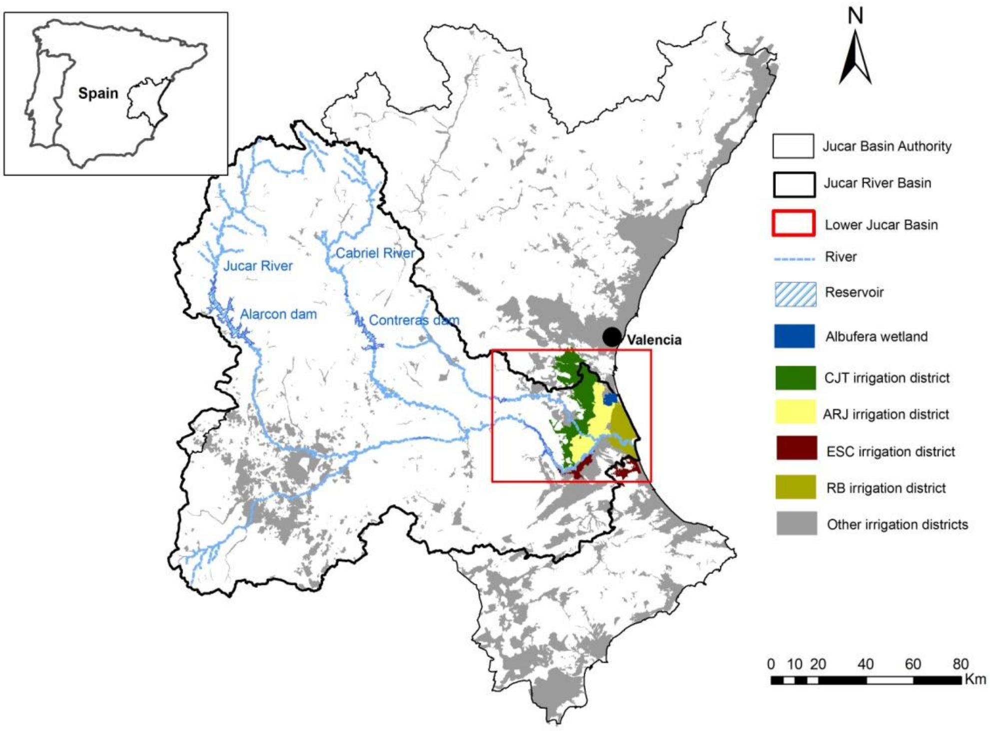
The paper is organized as follows. First, the lower Jucar basin is described in section 2, followed by the explanation of the modeling framework in section 3. Climate change and adaptation scenarios are presented in section 4, and the simulation results in section 5. Finally, section 6 concludes with the summary and policy implications.

4.2 Case study area: the lower Jucar basin

The lower part of the Jucar basin is located in the region of Valencia in Spain (Figure 4.1). This basin has an irregular Mediterranean hydrology, characterized by recurrent drought spells and normal years with dry summers. Irrigated area in the lower Jucar basin expands over 102,000 ha, representing half of the irrigated area in the basin.

The main crops grown are rice, corn, tomato, watermelon, peach, and citrus. Extractions for irrigation are about 980 Mm^3 per year, of which 770 are surface water and 210 are groundwater resources (CHJ, 2014). The analysis undertaken in this paper focuses on the irrigation activities in the four major irrigation districts in the lower Jucar basin: Acequia Real del Jucar (ARJ), Escalona-Carcagente (ESC), Ribera Baja (RB), and Canal Jucar-Turia (CJT).

Figure 4.1. Map of the lower Jucar basin.



These districts use almost 80 percent of total extractions in the lower Jucar basin.

The lower Jucar basin includes the Albufera wetland, which is one of the most important aquatic ecosystems in Southern Europe. The Albufera is catalogued in the RAMSAR list, and declared a special protected area for birds. It receives water mainly from the return flows of the ARJ and RB districts. Other flows originate from the neighboring Turia basin, and from the discharge of untreated and treated urban and industrial wastewaters in the adjacent municipalities.

The growth of water extractions in the upper Jucar and the severe drought spells in recent decades have triggered considerable negative environmental and economic impacts in the basin. For instance, water available to the ARJ district has been reduced from 700 to 200 Mm^3 in the last 40 years. Consequently, the dwindling irrigation return flows have caused serious environmental problems to the Albufera wetland. In addition, the outflows of the Jucar River to the Mediterranean Sea are below 1 m^3/s, which is very low compared with the other two major rivers in the region, the Ebro and Segura Rivers (Garcia-Molla et al., 2013).

One key issue for water management in the lower Jucar basin is adaptation of irrigation to the upcoming effects of climate change, which would exacerbate water scarcity and the intensity and frequency of droughts. Estimations of climate change impacts in the Jucar basin for a range of climate and socioeconomic scenarios for 2100 indicate a reduction of rainfall by up to 25 percent, an increase of temperature by up to 5 ºC, an increase of evapotranspiration by up to 22 percent, and a reduction of runoff by up to 45 percent (CEDEX, 2010).

4.3 Modeling framework

There is a growing body of economic literature that analyses climate change impacts and adaptation possibilities in irrigation. Two major approaches are widely used. One approach is mathematical programming models (both partial and general equilibrium models) that link biophysical (hydrologic, agronomic, and environmental) and economic components to simulate farmers' choices of crop mix, technologies, and resources for different climate scenarios, allocation rules, institutional arrangements, and policy interventions (Hurd et al., 2004; Connor et al., 2012; Medellin et al., 2013; Calzadilla et al., 2014). The alternative approach is econometric models that represent observed responses of farmers to past climate conditions under existing policies and institutions. These models are then used to estimate the effects of changes in climatic and policy variables (Zilberman et al., 2002; Mendelsohn and Dinar, 2003; Wheeler et al., 2013; Connor et al., 2014). Generally, mathematical programming models are computationally intensive, while econometric models are data intensive.

The modeling approach used in this paper is discrete stochastic programming (DSP). The advantage of using DSP models compared to other modeling techniques is their ability to capture sources of risk that influence the objective function and the constraint set, and also allowing for a multi-stage decision process in which the decision makers' knowledge about random events changes through time as economic choices are made (Rae, 1971). DSP has been previously used in many studies in the literature to analyze different water management problems. Some examples are the measurement of forgone irrigation benefits derived from rural to urban water transfers under uncertain water supplies (Taylor and Young, 1995), the impacts of reducing pumping in the Edwards aquifer in Texas (McCarl et al., 1999), and the assessment of water market outcomes under uncertain water supply in Spain (Calatrava and Garrido, 2005). DSP models seem to be a suitable approach to investigate irrigation adaptation to climate change because they can incorporate the production decisions in agriculture and the uncertainty linked to climate change impacts (Connor et al., 2012).

This paper develops a two-stage DSP framework. The first stage represents farmers' choice of long-run capital investment in cropping and irrigation systems. This investment is the response to the expected climate change scenario made prior to the knowledge on annual water inflows, which is a stochastic variable. The long-run horizon is given by the economic life of the capital investment which is in the range of 20 to 30 years. The second stage represents the short-run (annual) choice of variable input levels, including irrigated and fallowed areas, and irrigation water applied to crops which are determined after stochastic annual water inflows are known. This short-run choice is conditional on the fixed capital investment level chosen in the first stage.

The objective of the model is to maximize farmers' profits in each irrigation district subject to technical and resource constraints, which is given by the following formulation:

$$Max\ \Pi_k = \left[-\sum_i fcc_{i,k} - \sum_{i,j} fic_{i,j,k}\right] \cdot A1_{i,j,k} \tag{4.1a}$$

$$+\sum_s pr_s \cdot \sum_{i,j} p_i \cdot Y_{i,j,k,s}\left(W_{i,j,k,s}\right) \cdot A2_{i,j,k,s} \tag{4.1b}$$

$$-\sum_s pr_s \cdot \sum_{i,j} pw_k \cdot IW_{i,j,k,s} \cdot A2_{i,j,k,s} \tag{4.1c}$$

$$-\sum_s pr_s \cdot \sum_{i,j} vc_{i,k} \cdot A2_{i,j,k,s} \tag{4.1d}$$

$$-\sum_s pr_s \cdot \sum_{per,j} yp_{per,j,k} \cdot AF_{per,j,k,s} \tag{4.1e}$$

where variables are presented by capital letters. Π_k is farmers' profits in irrigation district k; $A1_{i,j,k}$ is the area of crop i equipped with irrigation system j in district k in the first stage; $A2_{i,j,k,s}$ is the irrigated area of crop i equipped with irrigation system j in district k and state of nature s in the second stage. $Y_{i,j,k,s}$ is yield of crop i equipped with irrigation system j in district k and state of nature s, which depends on the water applied to the crop, $W_{i,j,k,s}$. $IW_{i,j,k,s}$ is gross irrigation requirement of crop i equipped with irrigation system j in district k and state of nature s. $AF_{per,k,s}$ is the fallowed area of perennial crop, per ($per \subset i$), in district k and state of nature s.

Parameters are represented by lower case letters, where $fcc_{i,k}$ is fixed crop establishment costs; $fic_{i,j,k}$ is fixed irrigation equipment costs; p_i is crop prices; pw_k is water cost; $vc_{i,k}$ is variable cost other than water; $yp_{per,k}$ is perennial land fallowing penalty; and pr_s is the probability of each state of nature s.

The crops i which are included in the model are the main crops cultivated in the study area: rice, cereals, vegetables, citrus, and other fruit trees. The irrigation systems j are flood, sprinkler and drip. Surface water inflows to the basin in the period 1990-2011 are classified into four states of nature (s). The states are low, moderately low, moderately high, and high inflow levels, with probabilities of 10%, 40%, 40%, and 10%, respectively.

Expression (4.1a) represents long-run (first-stage) capital investment costs in cropping and irrigation systems. Expression (4.1b), (4.1c), and (4.1d) represent short-run (second-stage) crop revenues, water costs, and variable costs, respectively. Expression (4.1e) represents a perennial land fallowing penalty, indicating possible future yield losses if farmers decide to fallow perennial crop lands.

The yields, $Y_{i,j,k,s}$, are determined using crop-water production functions. These functions represent crop yield as an increasing function of water available for the crop up to a point beyond which additional water reduces yield. These quadratic production functions take the following form:

$$Y_{i,j,k,s}\left(W_{i,j,k,s}\right) = a_{i,j,k,s} + b_{i,j,k,s} \cdot W_{i,j,k,s} + c_{i,j,k,s} W_{i,j,k,s}^{2} \tag{4.2}$$

where the parameters a, b, and c are the intercept, linear and quadratic coefficients, respectively. These functions are estimated following the procedure developed by Warrick and Yates (1987) that relates crop yield to maximum and minimum crop water requirements and application uniformity. The production functions are calibrated based on local yield, water requirement, and economic data from chapter 1 of this thesis.

The variable applied water, $W_{i,j,k,s}$, is defined as the quantity of water available for each crop i equipped with irrigation system j in district k and state of nature s, which is the sum of net irrigation water and effective rainfall. This relationship is defined as follows:

$$W_{i,j,k,s} = IW_{i,j,k,s} \cdot ef_{j,k} + ER_{i,k,s} \tag{4.3}$$

where $ef_{j,k}$ is the efficiency of each irrigation system j in district k, and $ER_{i,k,s}$ is effective rainfall for each crop i in district k and state of nature s.

Crop-water production functions allow for the modeling of deficit irrigation or applying less than full crop water requirement and accepting less than the maximum possible yield, subject to a minimum water requirement threshold.

The objective function (1a-e) is maximized subject to the following constraints:

$$\sum_{i,j} A1_{i,j,k} \leq landavail_k \quad \forall\, k \tag{4.4}$$

$$A2_{i,j,k,s} \leq A1_{i,j,k} \quad \forall\, i,j,k,s \tag{4.5}$$

$$AF_{i,j,k,s} = A1_{i,j,k,s} - A2_{i,j,k,s} \quad \forall\, i,j,k,s \tag{4.6}$$

$$\sum_{i,j} IW_{i,j,k,s} \cdot A2_{i,j,k,s} \leq wateralloc_{k,s} \quad \forall\, k,s \tag{4.7}$$

$$\Phi_{k,s} = \left[wateralloc_{k,s} - \sum_{i,j} IW_{i,j,k,s} \cdot A2_{i,j,k,s}\right] \tag{4.8a}$$

$$+\left[\sum_{i,j} IW_{i,j,k,s} \cdot A2_{i,j,k,s} \cdot \left(1 - ef_{j,k}\right)\right] \quad \forall\, k,s \tag{4.8b}$$

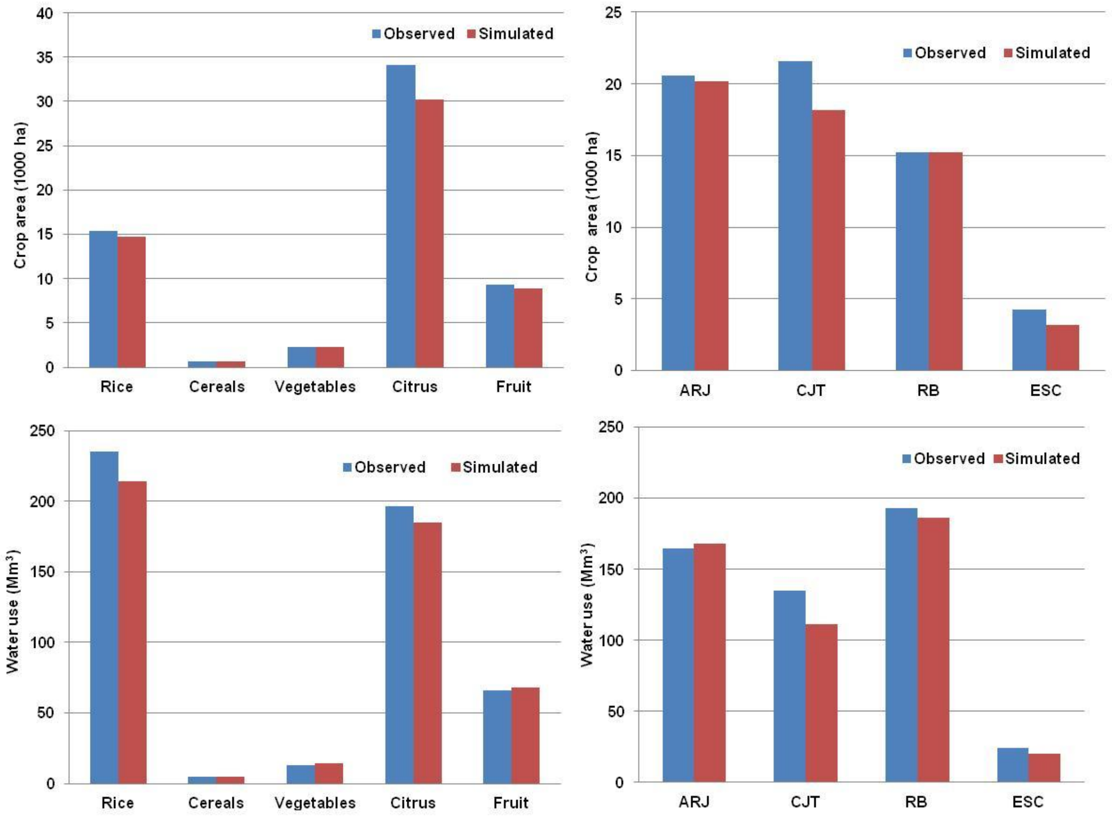
$$\Psi_s = \alpha \cdot \Phi_{ARJ,s} + \beta \cdot \Phi_{RB,s} \quad \forall s \tag{4.9}$$

Expression (4.4) represents land available in each irrigation district, $landavail_k$, for capital investments in cropping and irrigation systems (first-stage decision). Expressions (4.5) and (4.6) represent the possibility that a share of area with capital investments, $A1_{i,j,k}$, can be irrigated, $A2_{i,j,k,s}$, or fallowed, $AF_{i,j,k,s}$, in each state of nature (second-stage decision). Expression (4.7) states that the water used in an irrigation district under each state of nature does not exceed the water allocated to that district, $wateralloc_{k,s}$. Expression (4.8a-b) calculates irrigation water left for environmental flows in each irrigation district and state of nature, $\Phi_{k,s}$, which is the sum of unused irrigation water (4.8a), and irrigation return flows (4.8b). Irrigation return flows are calculated as a function of water use and efficiency. Expression (4.9) determines the quantity of water flowing to the Albufera wetland, the most important aquatic ecosystem in the Jucar basin, from environmental flows in each state of nature, Ψ_s. Parameters α and β represent the shares of environmental flows that feed the wetland from the ARJ and RB irrigation districts, respectively. $\Phi_{k,s}$ and Ψ_s are proxy variables for environmental impacts of climate change.

The environmental damage costs to the Albufera wetland from climate change are estimated indirectly. Given the limited knowledge and information available on ecosystem damages from the reduction of inflows to the wetland, a damage cost avoided method is used. This method does not provide a strict measure of ecosystem damages, but a lower-bound estimate of these damages (De Groot et al., 2002).

The hydrological plan of the Jucar basin indicates that the loss of inflows to the Albufera wetland under climate change would be replaced using available treated

Figure 4.2. Observed and simulated area and water use by crop and irrigation district.



wastewater. Therefore, the wastewater treatment cost is considered here as the damage avoidance cost for the Albufera wetland, which is estimated at 0.7 € per cubic meter of wetland inflow reduction relative to the current situation (CHJ, 2014).

Detailed information on the technical coefficients and parameters of the model have been collected from field surveys, expert consultation, statistical reports, and reviewing the literature. This information covers crop yields and prices, water and production costs, crop water requirements, irrigation efficiencies, and land availability (GV, 2009; INE, 2009; MARM, 2010).

The use of mathematical programming models to analyze agricultural production at regional level faces the problem of aggregation and overspecialization because farms in a region are different in terms of resources endowment, technologies, and management skills. Ideally, a regional model should include a component for every individual farm, but this is unfeasible because of the complexity of such a model (Hazell and Norton, 1986). Many approaches have been developed to solve this problem and to calibrate regional models to observed conditions such as the representative farm approach (Day, 1963), the convex combination approach (Önal and McCarl, 1991), and the positive

Table 4.1. Water allocation to irrigation districts by climate scenario and state of nature.

Climate scenario	State of nature	P value (%)	Water allocations (Mm^3)			
			ARJ	ESC	RB	CJT
Baseline	Low	10	61	12	119	17
	Moderately low	40	111	22	217	34
	Moderately high	40	168	38	274	54
	High	10	222	66	336	80
Climate change	Low	10	49	9	91	8
	Moderately low	40	89	16	166	16
	Moderately high	40	134	28	210	25
	High	10	178	49	257	37

mathematical programming (PMP) approach (Howitt, 1995; Röhm and Dabbert, 2003).

Our model is calibrated for the year 2009 (a moderately high state of nature year), with observed crop area, and water use by crop and irrigation district using the PMP approach. The Röhm and Dabbert's procedure is applied, in which there is a larger elasticity of substitution among crop variants than among completely different crops. Crop variants include the same crop grown under different irrigation systems. The outcomes of the model are broadly consistent, indicating that the model reproduces reliably the observed situation (Figure 4.2).

4.4 Climate change and adaptation scenarios

The modeling framework is used to analyze climate change impacts and adaptation possibilities in the Jucar basin. The impacts of an average climate change scenario are evaluated with a 32% reduction of water inflows to the basin, and a 15% increase of crop irrigation requirements compared to the baseline scenario (current climate conditions). These estimates of inflows and water requirements are taken from climate change projections of the Jucar basin by CEDEX (2010), which downscales to basin level the results of various global circulation models and emission scenarios.

Table 4.1 shows water allocations to irrigation districts under each climate scenario and state of nature. The allocations are estimated using the reduced form hydrological model of the Jucar basin developed in chapter 1 of this thesis. This model includes several demand nodes from upstream to downstream river reaches, and allocates water to those nodes subject to various hydrologic, institutional and environmental constraints.

Table 4.2. Climate change adaptation scenarios.

Adaptation possibilities	No-policy (NP)	Irrigation subsidy (IS)	Water market (WM)	Full adaptation (FA)
On-farm adaptation				
Crop mix change	Yes	Yes	Yes	Yes
Irrigation system change	Yes	Yes	Yes	Yes
Land fallowing	Yes	Yes	Yes	Yes
Deficit irrigation	Yes	Yes	Yes	Yes
Institutional adaptation				
Irrigation subsidy	No	Yes	No	Yes
Water trading	No	No	Yes	Yes

Four adaptation scenarios of several on-farm and institutional adaptation measures are analyzed. Adaptation measures at farm-level are crop mix and irrigation system change, land fallowing, and deficit irrigation. Adaptation measures at institutional-level are public subsidies for investments in efficient irrigation systems on-farm (sprinkler and drip systems), and introduction of water trading.

Table 4.2 shows the four adaptation scenarios, representing the different combinations of on-farm and institutional adaptation measures. Adaptation scenarios show the contribution of each measure to overall adaptation, and the tradeoff between the different possibilities. The objective function (4.1a-e) and the water availability constraint (4.7) are modified according to the adaptation scenario.

Farmers in scenarios are assumed to optimize the water application rate (deficit irrigation), which requires advanced technical skills for farmers, and available meteorological data and information on crop water requirements. A sensitivity analysis of this assumption is conducted by modeling the alternative assumption that farmers maintain fixed the water application rates.

4.5 Results and discussion

Results of the climate and adaptation scenarios are presented in terms of economic impacts, land use and irrigation system changes, and water use and environmental flows. Table 4.3 presents the economic outcomes of the various climate and adaptation scenarios, and figure 4.3 displays crop production costs, revenues, and profits per unit of land for each scenario.

Table 4.3. Economic outcomes of the climate and adaptation scenarios (10^6 €).

Economic indicators	Baseline	Climate change			
		NP	IS	WM	FA
Long-run fixed costs	120.1	87.9	96.9	100.7	108.7
Short-run variable costs*	93.2	65.8	73.3	82.9	87.8
Fallow penalty	1.6	0.5	0.5	0.0	0.0
Crop revenues	278.2	197.7	219.6	238.4	256.8
Public subsidy	0.0	0.0	4.6	0.0	4.9
Farmers' profits	63.3	43.4	48.8	54.8	60.2
Environmental costs	-	8.0	10.7	11.5	12.5

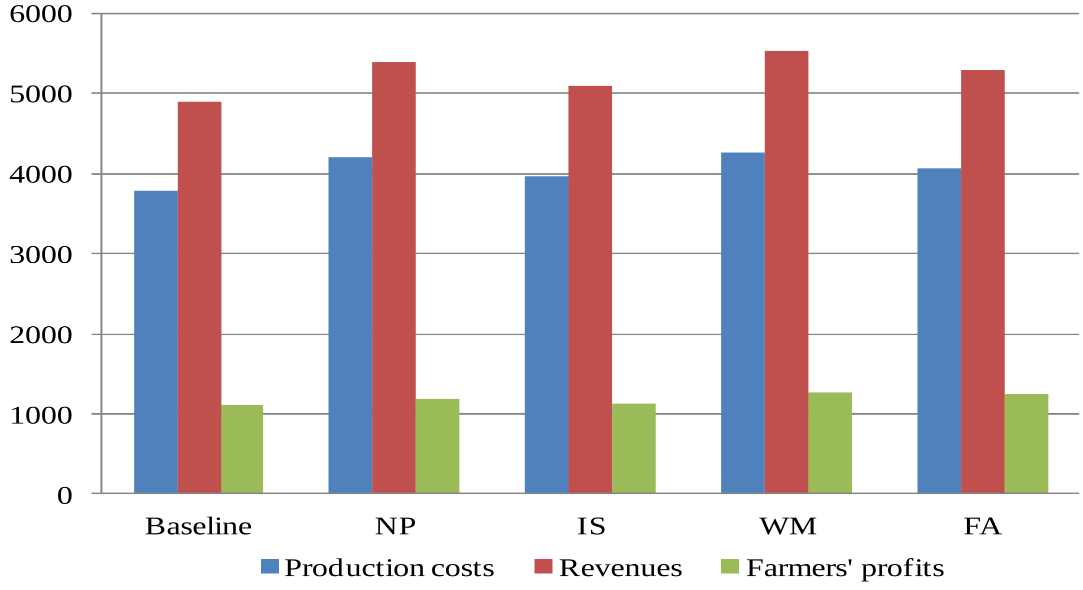
* Short-run variable costs include water and non-water costs.

Results indicate that climate change will likely have negative effects on irrigation activities in the Jucar basin for all scenarios considered. However, the severity of those effects is different depending on the scenarios. Farmers' profits are reduced by 31% under the most restrictive scenario (NP), and only by 5% under the most flexible scenario (FA) compared to baseline scenario. Introducing water trading (scenario WM) is the best individual adaptation option, improving farmers' profits by 26% (or 11 million €/year) compared to the most restrictive scenario. Subsidizing irrigation modernization (scenario IS) improves farmers' profits by 12% (or 5 million €/year). However, improved farmers' profits from irrigation subsidy barely cover the public subsidy cost. These results suggest that the extent of climate change impacts on irrigation will depend on government policy settings and farmers' adaptation responses.

Crop revenues and production costs (long and short-run) decrease under climate change for all scenarios compared to the baseline. However, they increase progressively as more adaptation possibilities are included. Production costs, revenues and profits per unit of land increase under all climate change and adaptation scenarios compared to the baseline scenario. The reason is that more water scarcity results in higher shadow values of water, inducing farmers to invest in efficient irrigation systems and high-value crops, and move-away from water-intensive and low-value crops.

The perennial land fallowing penalty arises from not meeting a minimum irrigation threshold for fruit trees that ensures productivity in future years. This penalty decreases under climate change compared to the baseline scenario, and vanishes under the water market scenarios (WM, FA). The environmental costs required to replace the water inflows losses to the Albufera wetland increase considerably under climate change for

Figure 4.3. Crop costs, revenues and profits by climate and adaptation scenarios (€/ha).



all scenarios considered, reaching almost 13 million €/year for the most flexible scenario.

The economic outcomes described above are explained by farmers' long-run choice of capital investment in cropping and irrigation systems (Table 4.4), and short-run choice of irrigated and fallowed areas (Table 4.5). Long-run choice of capital investment indicates that under climate change farmers reduce irrigated land between 15 and 35% compared to the baseline scenario. Irrigation subsidy and water market policies result in almost the same rate of irrigation abandonment (24%) compared to the baseline scenario.

The crop mix changes considerably, with a decline in the water-intensive and low-value crops, mainly rice, and the maintenance of high-value crops such as vegetables and fruit trees. Irrigated area falls by up to 65% for rice, 34% for cereals, and 39% for citrus, while the area of vegetables and other fruit trees remains almost unchanged. The reason for the large reduction in the area of citrus under climate change in some scenarios (NP, IS) is the lack of enough water in dry years (low water state of nature) to meet citrus minimum water requirements. Thus, the efficient response in the presence of substantial cultivated area of citrus is to reduce long-run capital investment to minimize both current and future yield losses. However, in the water market scenarios (WM, FA) more area of citrus is maintained because of the possibility of purchasing water in dry years to avoid future yield losses.

Table 4.4. Long-run choices by climate and adaptation scenario (ha).

Land use indicators	**Baseline**	**Climate change**			
		NP	**IS**	**WM**	**FA**
Irrigated land	56710	36660	43030	43035	48430
Land abandonment	0	20050	13680	13675	8280
Crop mix					
Rice	14740	6890	10085	5090	8260
Cereals	600	440	485	400	580
Vegetables	2310	2270	2290	2275	2265
Citrus	30170	18510	22170	26900	28260
Other fruit trees	8890	8550	8000	8370	9065
Irrigation system					
Flood	31980	24110	16975	32245	22770
Sprinkler	150	115	145	65	210
Drip	24580	12435	25910	10725	25450

Long-run choice of capital investment in irrigation systems suggests that farmers choose to move away from less-efficient flood system towards more-efficient sprinkler and drip systems in some scenarios (IS, FA). In these scenarios, the irrigation subsidy provides a good incentive to farmers for such a change. However contrary to expectations, farmers reduce the area under sprinkler and drip systems in the other scenarios (NP, WM). The main reason for that is the possibility of strategically adopting deficit irrigation and/or purchasing water in the market as contingencies in dry years, instead of investing in efficient irrigation systems with high sunk costs that may be needed only in dry years and not in wet years (high water state of nature with low water requirements and abundant water availability). These findings are consistent with the results from other studies dealing with irrigation technology adoption under uncertainty (Carey and Zilberman, 2002; Cai and Rosegrant, 2004).

Results of short-run choice of irrigated and fallowed areas suggest that farmers' response to risk (stochastic inflows) is similar in the various climate and adaptation scenarios. Famers mostly choose to irrigate areas which have strong capital investments in high-value crops (vegetables and fruit trees) and high-efficient irrigation technologies (sprinkler and drip), and to fallow areas with small capital investments in low-value crops (rice and cereals) and less-efficient irrigation technologies (flood). This behavior is especially pronounced in the water market scenarios (WM, FA) because of the possibility of water reallocation from crops with low to high marginal value of water.

Table 4.5. Short-run choices by climate and adaptation scenario (ha)*.

Land use indicators	Baseline	Climate change			
		NP	IS	WM	FA
Irrigated area	52720	35475	40220	41210	45230
Fallowed area	3990	1185	2810	1825	3200
Crop mix					
Rice	11185	5945	7520	3380	5125
Cereals	510	400	440	360	520
Vegetables	2180	2145	2160	2200	2260
Citrus	30025	18505	22165	26900	28260
Other fruit trees	8820	8480	7935	8370	9065
Irrigation system					
Flood	28110	23045	14290	30490	19595
Sprinkler	140	110	135	60	185
Drip	24470	12320	25795	10660	25450

* Results on short-run choice are average values across probability weighted states of nature.

However, the proportion of irrigated and fallowed areas is different for each scenario. The reason is the tradeoff between maximizing the expected profit and minimizing the risk of profit loss in dry years in each scenario. In the most restrictive scenario (NP), farmers seek to limit their risk exposure by reducing long-run investments and thus minimizing short-run losses of fallowing the cultivation area. But in the other scenarios (IS, WM, FA) farmers increase long-run investments, even if they have to fallow greater cultivation area because they can compensate dry years losses with higher gains in wet years, and also because they can purchase water in the market to offset the effects of drought.

Table 4.6 presents the water outcomes of the various climate and adaptation scenarios. The allocated water to irrigation is divided between water used by crops and unused water left in-stream. Water use under climate change decreases by up to 23% compared to the baseline scenario, although water use increases progressively as more adaptation options are included. Water use expands by 7% under the most flexible scenario (FA) compared to the most restrictive scenario (NP). Water use increases by 6% with the irrigation subsidy, and by 3% with water trading. Both the irrigation subsidy and water trading seem to provide significant incentives for farmers to use the water allocations that are left in-stream in wet years under the most restrictive scenario. These water allocations become activated by expanding the irrigated area of flexible annual crops in wet years. The in-stream unused water is reduced by up to 33% compared to the most restrictive scenario.

Table 4.6. Water outcomes by climate and adaptation scenarios (Mm^3)*.

Water indicators	Baseline	Climate change			
		NP	IS	WM	FA
Water use	449	347	367	358	373
Unused water	94	78	58	67	53
Environmental flows	217	174	146	164	136
Inflows to Albufera	45	33	29	28	27

* Results of the water indicators are average values across probability weighted states of nature.

The decrease of both the volume of water left in-stream and the irrigation return flows leads to a reduction of environmental flows by up to 37% compared to the baseline scenario. The consequence is a fall of the inflows to the Albufera wetland by up to 40% compared to the baseline scenario. The irrigation subsidy contributes more to the reduction of environmental flows, followed by water trading. However, water trading contributes somewhat more to the reduction of inflows to the Albufera wetland than irrigation subsidy. The reason is the market spatial reallocation of water and the fact that the Albufera is fed by the return flows of the ARJ and RB irrigation districts which are net water sellers in the market (low-value uses).

Table 4.7 presents the results of the sensitivity analysis of the deficit irrigation assumption in terms of farmers' profits, environmental costs, long-run choices, and water outcomes. Results show that without deficit irrigation farmers' profits for the various climate and adaptation scenarios are further reduced compared to baseline scenario as a result of less capacity to maintain irrigated area by reducing water application rates on low-value crops. The long-run choice of capital investment in cropping and irrigation systems decreases by up to 16%, and land abandonment increases by up to 94% compared to the same scenarios with the possibility of deficit irrigation. These results highlight the extent to which farmers are able to perform potential adaptation opportunities at farm level.

Water use decreases under all adaptation scenarios compared to the baseline scenario and to adaptation scenarios with the possibility of deficit irrigation. The elimination of the deficit irrigation possibility leads to the fall of water use and an increase of the inflows to the Albufera. The irrigation subsidy remains the largest contributor to environmental degradation. Environmental costs are reduced compared to adaptation scenarios with the possibility of deficit irrigation.

Table 4.7. Results of the sensitivity analysis.

	NP	IS	WM	FA
Farmers' profits (10^6 €)	39.5	43.5	47.7	51.8
Environmental costs (10^6 €)	7.2	9.5	8.3	9.4
Long-run irrigated land (ha)	33592	38523	38901	40679
Long-run land abandonment (ha)	23114	18183	17805	16027
Water use (Mm^3)	324	353	351	359
Environmental flows (Mm^3)	172	152	157	148
Inflows to Albufera (Mm^3)	34	31	33	31

4.6 Conclusions and policy implications

This paper presents a comparative analysis of the effectiveness of two popular incentive-based water management policies to address climate change impacts on irrigated agriculture in Southern Europe: water markets and irrigation subsidies. The analysis is undertaken in a representative basin of Southern Europe, the lower Jucar basin of Spain, using a modeling framework that links hydrologic, agronomic, and economic variables within a discrete stochastic programming model. This model estimates farmers’ responses to climate change and policy interventions in terms of long-run choices of capital investment in cropping and irrigation systems and short-run decision to irrigate or fallow land.

Results indicate that climate change will likely substantially reduce farmers’ profits in the absence of any policy intervention. These losses can be reduced through the implementation of water markets and irrigation subsidy policies. These policies provide incentives to farmers for investing in cropping and irrigation systems, reducing land abandonment, shifting towards high-value cultivation activities, and increasing water use, although farmers’ behavior is different under each policy. In addition, a deficit irrigation strategy proves to be an important response to climate change, reducing significantly farmers’ losses. However, environmental flows will be reduced under climate change for all scenarios considered, generating considerable environmental costs for society. Water market and irrigation subsidy policies further reduce environmental flows compared to a climate change scenario without any policy intervention, with larger flow reductions from irrigation subsidies than water markets.

These empirical results suggest that the benefits of the irrigation subsidy policy are very small, especially when public subsidies and social costs of replacing lost environmental flows are accounted for. In contrast, the benefits of introducing water markets seem to be quite large, even though well-functioning water markets involve sizeable monitoring and transaction costs that are not considered in this study but require evaluation.

As a final remark, the findings in the lower Jucar basin highlight that climate change will likely have negative impacts on the irrigated agriculture and the linked water-dependent ecosystems of Southern Europe. However, the severity of these impacts will depend on the degree of adaptation at farm level, farmers' investment decisions, and the policy choices, which are interrelated. Therefore, the main thrust of the European water and agricultural regulations should be placed on enhancing the adaptive capacity at farm level, improving farmers' knowledge of climate change impacts for better long-run investment decisions, and fostering the adoption of adaptation policies that minimize both private and social benefit losses such as water markets.

4.7 References

Blewett, R., 2012. Shaping a Nation. A Geology of Australia. Geoscience Australia-ANU Press. Canberra.

Booker, J., Michelsen, A., Ward, F., 2005. Economic impacts of alternative policy responses to prolonged and severe drought In the Rio Grande Basin. *Water Resour. Res.* 4, 1-15.

Cai, X., Rosegrant, M., 2004. Irrigation technology choices under hydrologic uncertainty: A case study from Maipo River Basin, Chile. *Water Resour. Res.* 40, 1-10.

Calatrava, J., Garrido, A., 2005. Modelling water markets under uncertain water supply. *Eur. Rev. Agric. Econ.* 32 (2), 119-142.

Calzadilla, A., Zhu, T., Rehdanz, C., Tol, R., Ringler, C., 2014. Climate change and agriculture: Impacts and adaptation options in South Africa. Water Resources and Economics 5, 24-48.

Carey, J., Zilberman, D., 2002. A model of investment under uncertainty: modern irrigation technology and emerging markets in water. *Am. J. Agr. Econ.* 84 (1), 171-183.

Cazcarro, I., Duarte, R., Sanchez, J., Sarasa, C., 2011. Water rates and the responsibilities of direct, indirect and end-users in Spain. *Economic Systems Research* 23 (4), 409-430.

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.

Connor, J., Schwabe, K., King, D., Knapp, K., 2012. Irrigated agriculture and climate change: the influence of water supply variability and salinity on adaptation. *Ecol. Econ.* 77, 149-157.

Connor, J., Kaczan, D., 2013. Principles for economically efficient and environmentally sustainable water markets: The Australian experience. In: Schwabe, K., Albiac, J., Connor, J., Hassan, R., Meza, L., (Eds.), Drought in Arid and Semi-Arid Environments: A Multi-Disciplinary and Cross-Country Perspective. Springer, Dordrecht, pp 357-374.

Connor, J., Kandulu, J., Bark, R., 2014. Irrigation revenue loss in Murray–Darling Basin drought: An econometric assessment. *Agr. Water Manage.* 145, 163-170.

Confederación Hidrográfica del Júcar (CHJ), 2014. Memoria proyecto del plan hidrológico de cuenca. MAGRAMA. Valencia.

Day, R., 1963. On aggregating linear programming models of production. *J. Farm Econ.* 45, 797-813.

De Groot, R., Wilson, M., Boumans, R., 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecol. Econ.* 41, 393-408.

Elliot, J., Deryng, D., Müller, C., et al., 2014. Constraints and potentials of future irrigation water availability on agricultural production under climate change. *PNAS* 111(9), 3239-3244.

European Commission (EC), 2009. Common implementation strategy for the Water Framework Directive (2000/60/EC). Guidance document nº 24, river basin management in a changing climate. Technical report 2009-040. Brussels.

European Commission (EC), 2013. Principles and recommendations for integrating climate change adaptation considerations under the 2014-2020 programmes. SWD 139 final. Brussels.

European Environmental Agency (EEA), 2012. Climate change, impacts and vulnerability in Europe 2012: an indicator-based report. EEA report nº 12. EEA. Copenhagen.

Garcia-Molla, M., Sanchis, C., Ortega, M., Avella, L., 2013. Irrigation Associations Coping with Drought: The Case of Four Irrigation Districts in Eastern Spain. In: Schwabe, K., Albiac, J., Connor, J., Hassan, R., Meza, L., (Eds.), Drought in Arid and Semi-Arid Environments: A Multi-Disciplinary and Cross-Country Perspective. Springer, Dordrecht, pp 101-122.

Gohar, A., Ward, F., 2010. Gains from expanded irrigation water trading in Egypt: An integrated basin approach. *Ecol. Econ.* 69, 2535-2548.

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.

Gomez-Limon, J., Martinez, Y., 2006. Multi-criteria modelling of irrigation water market at basin level: A Spanish case study. *Eur. J. Oper. Res.* 173, 313-336.

Graveline, N., Majone, B., Van Duinen, R., Ansink, E., 2013. Hydro-economic modeling of water scarcity under global change: an application to the Gállego river basin (Spain). *Reg. Environ. Change* 14(1), 119-132.

Hazell, P., Norton, R., 1986. Mathematical programming for economic analysis in agriculture. Macmillan Publishing Company, New York.

Howe, C., Schurmeier, D., Shaw, W., 1986. Innovative Approaches to Water Allocation: The Potential for Water Markets. *Water Resour. Res.* 22(4): 439-449.

Howden, M., Soussana, J., Tubiello, F., Chhetri, N., Dunlop, M., Meinke, H., 2007. Adapting agriculture to climate change. *PNAS* 104 (50), 19691-19696.

Howitt, R., 1995. Positive mathematical programming. *Am. J. Agr. Econ.* 77: 329-342.

Huffaker, R., 2008. Conservation potential of agricultural water conservation subsidies. *Water Resour. Res.* 44, 1-8.

Hurd, B., Callaway, M., Smith, J., Kirshen, P., 2004. Climatic change and U.S. water resources: from modeled watershed impacts to national estimates. *J. Am. Water Resour. As.* 40 (1), 129-148.

Intergovernmental Panel on Climate Change (IPCC), 2014. Summary for Policymakers. In: Field, C., Barros, V., Dokken, D., et al. (Eds.), Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press. Cambridge.

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

Jimenez, B., Oki, T., Arnell, N., et al., 2014. Freshwater resources. In: Field, C., Barros, V., Dokken, D., et al. (Eds.), Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press. Cambridge.

Leclere, D., Jayet, P., de Noblet-Ducoudre, N., 2013. Farm-level autonomous adaptation of European agricultural supply to climate change. *Ecol. Econ.* 87, 1-14.

Lehner, B., Döll, P., Alcamo, J., Henrichs, T., Kaspar, F., 2006. Estimating the impact of global change on flood and drought risks in Europe: A continental, integrated analysis. *Climatic Change* 75, 273-299.

McCarl, B., Dillion, C., Keplinger, K., Williams, R., 1999. Limiting pumping from the Edwards Aquifer: an economic investigation of proposal, water markets, and spring flow guarantees. *Water Resour. Res.* 34(4), 1257-1268.

Medellin, J., Howitt, R., Lund, J., 2013. Modeling Economic-Engineering Responses to Drought: The California Case. In: Schwabe, K., Albiac, J., Connor, J., Hassan, R., Meza, L., (Eds.), Drought in Arid and Semi-Arid Environments: A Multi-Disciplinary and Cross-Country Perspective. Springer, Dordrecht, pp 341-356.

Mendelsohn, R., Dinar, A., 2003. Climate, Water, and Agriculture. *Land Econ.* 79 (3), 328-341.

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.

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

Perry, C., Wichelns, D., Steduto, P., 2014. The myth that "water efficiency" will eradicate hunger and poverty. *Waterfront* 4, 10.

Playan, E., Lecina, S., Isidoro, D., Aragues, R., Faci, J., Salvador, R., Zapata, N., Cavero, J., 2013. Living with drought in the irrigated agriculture of the Ebro basin (Spain): structural and water management actions. In: Schwabe, K., Albiac, J., Connor, J., Hassan, R., Meza, L., (Eds.), Drought in Arid and Semi-Arid Environments: A Multi-Disciplinary and Cross-Country Perspective. Springer, Dordrecht, pp 101-122.

Qureshi, M., Schwabe, K., Connor, J., Kirby, M., 2010. Environmental water incentive policy and return flows. *Water Resour. Res.* 46, 1-12.

Rae, A., 1971. An empirical application and evaluation of discrete stochastic programming in farm management. *Am. J. Agr. Econ.* 53, 625-38.

Reidsma, P., Ewert, F., Lansink, A., Leemans, R., 2010. Adaptation to climate change and climate variability in European agriculture: the importance of farm level responses. *Eur. J. Agron.* 32, 91-102.

Röhm O. and S. Dabbert (2003). Integrating agri-environmental programs into regional production models: an extension of positive mathematical programming. *Am. J. Agr. Econ.* 85 (1), 254-265.

Taylor, R., Young, R., 1995. Rural-to-urban water transfers: measuring direct foregone benefits of irrigation water under uncertain water supplies. *J. Agr. Resour. Econ.* 20(2), 247-262

Varela, C., Blanco, I., Esteve, P., Bharwani, S., Fronzek, S., Downing, T., 2014. How can irrigated agriculture adapt to climate change? Insights from the Guadiana Basin in Spain. *Reg. Environ. Change*, December 2014.

Warrick, A., Yates, S., 1987. Crop yield as influenced by irrigation uniformity. In: Hillel D. (Ed.). Advances in Irrigation 4. Academic Press, New York.

Wheeler, S., Zuo, A., Bjornlund, H., 2013. Farmers' climate change beliefs and adaptation strategies for a water scarce future in Australia. *Global Environ. Chang.* 23, 537-547.

Wheeler, S., Loch, A., Zuo, A., Bjornlund, H., 2014. Reviewing the adoption and impact of water markets in the Murray-Darling Basin, Australia. *J. Hydrol.* 518, 28-41.

Zilberman, D., Dunarm, A., MacDougal, N., Khanna, M., Brown, C., Castillo, F., 2002. Individual and institutional responses to the drought: the case of California agriculture. *Journal of Contemporary Water Research and Education* 121, 17-23.

Chapter 5

Hydro-economic modeling with aquifer-river interactions for sustainable basin management

Abstract

Water demands for irrigation, urban and environmental uses in many arid and semiarid regions continue to grow, while freshwater supplies from surface and groundwater resources are becoming scarce and are expected to decline because of climate change. Policymakers in these regions are faced with hard choices on water management and policies. Hydro-economic modeling is the state-of-the arts tool to assist policymakers in the design and implementation of sustainable water management policies in basins. The strength of hydro-economic modeling lies in its capacity to integrate key biophysical and socio-economic components within a coherent framework. A major gap in developments of hydro-economic models to date has been the difficulty of integrating surface and groundwater flows based on the theoretically correct Darcy equations used by the hydrogeological community. The hydro-economic model presented here specifies a spatially-explicit groundwater flow element. The methodological contribution to previous modeling efforts is the explicit specification of the aquifer-river interactions, which are important when aquifer systems make a sizable contribution to basin resources. This advanced framework is applied to the Jucar basin (Spain) for the assessment of different climate change scenarios and policy choices, specially the hydrologic, land use and economic outcomes. The response to scenarios integrates the multiple dimensions of water resources, allowing results to provide valuable information on the basin scale climate change adaptation paths to guide alternative policy choices using sound science.

Keywords. Hydro-economic modeling, aquifer-river interactions, climate change, water policies

5.1 Introduction

Water resources are key critical assets to support human societies and natural ecosystems. Despite their paramount importance, many freshwater systems are threatened because of the affordable expansion in water extractions, coupled with large pollution loads that impair water quality. The rate of growth of water extractions has almost doubled population growth during the last century. This expanded human access to water has been driven by urbanization, industrialization and land use changes, with a large deployment of engineering waterworks such as dams, irrigation schemes, inter-basin transfers, and extensive well drilling.

Costs to ensuing damages to ecosystems and biodiversity in river basins are undervalued by private markets when there are public good characteristics of these natural assets. The environmental benefits and services provided to society are “market externalities” and the market failure that are corrected with water policies and regulations can produce a more economically efficient allocation of water resources (Dasgupta and Heal, 1979).

The situation in river basins located in arid and semiarid regions is even worse because in these regions human activities already maximize the extraction of water from the natural environment. The water scarcity problem could become quite serious, threatening both human activities and natural ecosystems. The forthcoming impacts from climate change would further exacerbate the current water scarcity situation in arid and semiarid regions having sizable impacts on irrigated agricultural production, as indicated by global model results (Schewe et al., 2014; Elliot et al., 2014).

The current drought in California and much of the southwestern United States and the recent millennium drought in the Murray-Darling basin of Australia illustrate vividly the severity of water scarcity problems. Another indicator is the finding that a third of the world biggest groundwater systems are in distress, especially in arid and semiarid basins (Richey et al., 2015). The long-term sustainability of groundwater systems requires new aquifer management models in order to address the current groundwater management challenge (Gorelick and Zheng, 2015).

This widespread mismanagement of water resources in basins demonstrate the need for better analytical tools that could support more sustainable water management and policies. An important emerging tool for the analysis of sustainable management options in basins is hydro-economic modeling. Hydro-economic models (HEM) integrate the spatially distributed water systems, water storage and conveying infrastructures, water-based economic activities, and water-dependent ecosystems into a coherent model (Harou et al., 2009). The advantage of this approach is the inclusion of interrelationships between the hydrologic, economic, institutional and environmental components for an accurate assessment of sustainable management and policy options (Cai et al., 2003). Booker et al. (2012) analyze the evolution in concepts, methods and application of hydro-economic modeling, stressing its capability for addressing system wide impacts. They indicate that hydro-economic modeling requires further advances in the dynamic and stochastic model dimensions, and also in the accurate understanding of interdependencies between the hydrologic, economic, institutional and environmental components. Despite these achievements, an important gap not yet closed in the development of most hydro-economic models is the theoretically weak connection of the linkages between groundwater and surface water activities. While the Darcy equation approach is the widely-recognized and correct approach to modeling groundwater flows, few if any hydro-economic modeling applications in the water resources literature properly account for the Darcy equation approach for groundwater, mass balance for surface water, and economic principles properly applied for a complete optimization framework.

This paper's unique contribution is to present the development and application of a hydro-economic modeling framework that addresses the gap described above. The issue addressed in this paper is the improvement of the river basin dynamics in modeling, by including the linkage between aquifer systems and river flows. This linkage is important when aquifer systems are closely related to river flows making a sizable inflow or outflow contribution to the basin resources. Overall, the aquifer dynamics and stream-aquifer interactions have been simplified in hydro-economic models, given the level of complexity already involved in modeling whole river basins. First, aquifers are represented as simple single-tank units. Second, the linkage of aquifers and river flows, either inflows or outflows, is usually represented with linear estimates relating the stream-aquifer flow, with variables such as aquifer recharge, water pumping, or water table levels.

For example, Cai et al. (2003) assume a linear relationship between aquifer discharges and water table levels. McCarl et al. (1999) use regression-based forecasts of aquifer discharges that respond to recharge, pumping and water table heights. Ward and Pulido-Velazquez (2009) estimate discharge using a simple proportion of recharge. The study by Schoups et al. (2006) deals with the conjunctive use of surface water and groundwater in irrigation, stressing the need to account for interactions between surface and groundwater systems. Although the model includes water extractions and returns to the aquifer from irrigation, it does not include an explicit aquifer-river interaction.

The approach to the aquifer-river interaction taken here is much more elaborated, avoiding both the single-tank assumption, and overly simple assumptions on the aquifer-river linkages. When these linkages are important, these simplifying assumptions may result into wrong policy recommendations (Brozovic et al., 2010). The groundwater flow formulation used in this paper is similar to the one used in MODFLOW groundwater model, which is able to simulate the spatial and temporal heterogeneity of real-world aquifers and the linkage between aquifer system and river flow (McDonald and Harbaugh, 1984). The model is applied to the Jucar basin in Spain, where the river-aquifer linkage is important for the sustainable management of the basin.

5.2 Modeling framework

This paper presents an integrated basin-scale hydro-economic modeling framework that could be used to assess the impacts of future climate change scenarios and to analyze the economic and biophysical outcomes of adaptation policies. This framework is a comprehensive tool that integrates several components including surface and groundwater hydrology, agronomy, land use, institutions, environment, and economic activities, covering the main water uses. The mathematical formulation of each component is presented below.

5.2.1 Hydrology

Basin hydrology is based on the principle of water mass balance, defined for each flow, i, and each stock, s. The main flow variables, X_i, tracked by the model include headwater flow, streamflow, surface water diversion, groundwater pumping, water applied and consumed, return flow to streams and aquifers, stream-aquifer interaction, and reservoir release and evaporation. The stock variables, Z_s, tracked by the model are the reservoir and aquifer volume levels.

5.2.1.1 Headwater inflows

Total surface water inflows to the basin are defined as the total annual flows at the different headwater gauges. The inflows, $X_{h,t}$, at each headwater gauge, h (a subset of i), in time t, are equal to total source supplies, $source_{h,t}$:

$$X_{h,t} = source_{h,t} \tag{5.1}$$

5.2.1.2 Streamflows

The streamflow, $X_{v,t}$, at each river gauge, v (a subset of i), in time t, is equal to the sum of flows over any upstream node, i, whose activities impact that streamflow. These nodes include headwater inflow, river gauge, diversion, surface return flow, stream-aquifer interaction, and reservoir release. The streamflow at each river gauge, which is required to be nonnegative, is defined as follows:

$$X_{v,t} = \sum_i B_{i,v} \cdot X_{i,t} \tag{5.2}$$

where $B_{i,v}$ is a vector of coefficients that links flow nodes, i, to river gauge nodes, v. The coefficients take on values of 0 for non-contributing nodes, +1 for nodes that add flow, and -1 for nodes that reduce flow.

5.2.1.3 Surface water diversions

Water supply to basin's users can be met partially or totally by diversions from a stream. However, during drought spells, streamflow can be low or even zero. Therefore, a surface water diversion constraint is required in order to avoid that diversion, $X_{d,t}$, exceeds available streamflow at each diversion node, d (a subset of i), in time t. A diversion, which is required to be nonnegative, is defined as follows:

$$X_{d,t} \leq \sum_i B_{i,d} \cdot X_{i,t} \tag{5.3}$$

where $B_{i,v}$ is a vector of coefficients that links flow nodes, i, to diversion nodes, d. The right hand side term represents the sum of all contributions to flow at diversion nodes from upstream sources. These sources include headwater inflow, river gauge, diversion, surface return flow, stream-aquifer interaction, and reservoir release. The B coefficients, take on values of 0 for non-contributing nodes, +1 for nodes that add flow, and -1 for nodes that reduce flow.

5.2.1.4 *Water applied*

Water applied, $X_{a,t}$, at each application node, a (a subset of i), in time t, can come from two sources: stream diversion, $X_{d,t}$, and groundwater pumping, $X_{p,t}$. Water applied is defined as follows:

$$X_{a,t} = \sum_{d} B_{d,a} \cdot X_{d,t} + \sum_{p} B_{p,a} \cdot X_{p,t} \tag{5.4}$$

where $B_{d,a}$ and $B_{p,a}$ are vectors of coefficients that link application nodes to diversion and pumping nodes, respectively. The coefficients take on values of 1 for application nodes withdrawing water from available sources, and 0 for not withdrawing water.

For each agricultural node in the basin, total water applied for irrigation is defined as follows:

$$X_{a,t}^{ag} = \sum_{j,k} B_{a,j,k} \cdot \left(\sum_{u} B_{u,a} \cdot L_{u,j,k,t}\right) \tag{5.5}$$

Equation (5.5) states that irrigation water applied to crops from both surface and groundwater sources, $X_{a,t}^{ag}$, is equal to the sum over crops (j) and irrigation technologies (k) of water application per ha, $B_{a,j,k}$, multiplied by irrigated area, $L_{u,j,k,t}$, for each crop and irrigation technology. $L_{u,j,k,t}$ is multiplied by an identity matrix, $B_{u,a}$, to conform nodes.

5.2.1.5 *Water consumed*

Consumptive use, $X_{u,t}$, at each use node, u (a subset of i), in time, t, is an empirically determined proportion of water applied, $X_{a,t}$. For irrigation, consumptive use is the amount of water used through crop evapotranspiration (ET). For urban uses, consumptive use is the proportion of urban water supply not returned through the sewage system. That use, which cannot be negative, is defined as follows:

$$X_{u,t} = \sum_{a} B_{a,u} \cdot X_{a,t} \tag{5.6}$$

where parameters, $B_{a,u}$, are coefficients indicating the proportion of water applied that is consumptively used in each use node. For agricultural use nodes, water consumed is measured as:

$$X_{u,t}^{ag} = \sum_{j,k} B_{u,j,k} \cdot L_{u,j,k,t} \tag{5.7}$$

Equation (5.7) states that irrigation water consumed, $X_{u,t}^{ag}$, is equal to the sum over crops (j) and irrigation technologies (k) of empirically estimated ET per ha, $B_{u,j,k}$, multiplied by irrigated area, $L_{u,j,k,t}$, for each crop and irrigation technology.

5.2.1.6 Return flows

Return flows, $X_{r,t}$, at each return flow node, r (a subset of i), in time, *t*, is a proportion of water applied $X_{a,t}$. These flows return to the river system or contribute to aquifers recharge. Return flows are defined as follows:

$$X_{r,t} = \sum_a B_{a,r} \cdot X_{a,t} \tag{5.8}$$

where parameters, $B_{a,r}$, are coefficients indicating the proportion of total water applied that is returned to river and aquifers. For agricultural nodes, returns flows are defined as follows:

$$X_{r,t}^{ag} = \sum_{j,k} B_{r,j,k} \cdot \left(\sum_u B_{u,r} \cdot L_{u,j,k,t}\right) \tag{5.9}$$

Equation (5.9) states that irrigation return flows, $X_{r,t}^{ag}$, are equal to the sum over crops (j) and irrigation technologies (k) of empirically estimated return flows per ha, $B_{r,j,k}$, multiplied by irrigated area, $L_{u,j,k,t}$, for each crop and irrigation technology. $L_{u,j,k,t}$ is multiplied by an identity matrix, $B_{u,r}$, to conform nodes. The sum of water consumed and returned must be equal to water applied at each demand node.

5.2.1.7 Reservoir stock and operation

Water stock, $Z_{res,t}$, at each reservoir, res (a subset of s), in time *t*, is defined in the following equations:

$$Z_{res,t} = Z_{res,t-1} - \sum_L B_{L,res} \cdot X_{L,t} - \sum_e B_{e,res} \cdot X_{e,t} \tag{5.10}$$

$$Z_{res,0} = B_{res,0} \tag{5.11}$$

$$Z_{res,t} \leq C_{res}^{max} \tag{5.12}$$

$$Z_{res,t} \geq C_{res}^{min} \tag{5.13}$$

where equation (5.10) states that reservoir water stock, $Z_{res,t}$, is equal to its stock in the previous time period, $Z_{res,t-1}$, minus both the net release (outflow minus inflow) from the reservoir, $X_{L,t}$, and reservoir evaporation, $X_{e,t}$. Evaporation depends on reservoir features and climatic factors. Both sets of parameters $B_{L,res}$ and $B_{e,res}$ are identity matrices linking reservoir stock nodes to reservoir release and evaporation nodes, respectively. Equation (5.11) defines initial reservoir water stock at $t = 0$, $B_{res,0}$. Upper and lower bounds on reservoir water stock are defined in equation (5.12) and (5.13), respectively. Parameters C_{res}^{max} and C_{res}^{min} are reservoir maximum capacity and dead storage, respectively. Upper bound constraint guarantees that reservoir stock in each time period never exceeds its maximum capacity, while lower bound constraint states the capacity from which stored water in reservoir cannot be used.

5.2.1.8 Aquifer stock and stream-aquifer interaction

The groundwater flow is calculated with a finite-difference groundwater flow equation based on the principle of water mass balance and Darcy's law. The formulation is similar to that used in MODFLOW groundwater model (McDonald and Harbaugh, 1984). Assume an aquifer system divided into n (1 row and n columns) connected cells (sub-aquifers), aqf (a subset of s), which are linked to n connected reaches of a river, $river$ (a subset of i). The aquifer head, $H_{aqf,t}$, in each sub-aquifer, aqf, in time, t, is defined in the following equation (see the mathematical appendix for further details on the groundwater flow equation):

$$H_{aqf,t} = \left[1/\left\{\left(S_{aqf} \cdot A_{aqf}/\Delta t\right) + C_{aqf,aqf-1} + C_{aqf,aqf+1} + C_{river,aqf}\right\}\right] \cdot \left[R_{aqf,t} - Q_{aqf,t} + \left(S_{aqf} \cdot A_{aqf} \cdot H_{aqf,t-1}/\Delta t\right) + C_{aqf-1} \cdot H_{aqf-1,t} + C_{aqf+1} \cdot H_{aqf+1,t} + C_{river,aqf} \cdot H_{river,aqf}\right] \;;\quad H_{aqf,0} = B_{aqf,0} \tag{5.14}$$

where parameters S_{aqf}, A_{aqf}, and $R_{aqf,t}$ are specific yield, area, and recharge for sub-aquifer, aqf, respectively. Parameters $C_{aqf,aqf-1}$ and $C_{aqf,aqf+1}$ represent hydraulic conductance between sub-aquifer, aqf, and adjacent sub-aquifers, $aqf - 1$ and $aqf + 1$, respectively. Parameter $C_{river,aqf}$ is hydraulic conductance of river reach, $river$, linked to sub-aquifer, aqf. Parameter Δt is the time step. Parameter $B_{aqf,0}$ is the initial head of sub-aquifer, aqf, at $t = 0$. Variable $H_{aqf,t-1}$ is the head of sub-aquifer, aqf, in the previous time period. Variables $H_{aqf-1,t}$ and $H_{aqf+1,t}$ are heads of adjacent sub-aquifers, $aqf - 1$ and $aqf + 1$, respectively. Variable $H_{river,aqf}$ is the head of the river reach, $river$, linked to sub-aquifer, aqf, and variable $Q_{aqf,t}$ is net groundwater pumping from sub-aquifer, aqf, which are defined in equations (5.15) and (5.16) as follows:

$$H_{river,aqf} = BH_{river,aqf} \cdot (\textstyle\sum_v B_{v,river,aqf} \cdot X_{v,t}) \tag{5.15}$$

$$Q_{aqf,t} = \sum_p B_{p,aqf} \cdot X_{p,t} - \sum_r B_{r,aqf} \cdot X_{r,t} \tag{5.16}$$

where variables $X_{v,t}$ is streamflow at each river gauge node, v; $X_{p,t}$ is gross groundwater pumping at each pumping node, p; and $X_{r,t}$ is return flows at each return flow node, r, in time, t. Parameters $BH_{river,aqf}$ are coefficients defining the relationship between river head (or river stage) and streamflow (or discharge) for each river reach. This relationship depends on river features such as riverbed form and roughness coefficients. Parameter sets $B_{v,river,aqf}$, $B_{p,aqf}$ and $B_{r,aqf}$ are identity matrices linking river reaches to river gauge nodes, and sub-aquifers to pumping and return flow nodes, respectively.

The interaction between each sub-aquifer and the corresponding river reach is defined in the following equation:

$$X_{river,aqf,t} = C_{river,aqf} \cdot \left(H_{river,aqf,t} - H_{aqf,t}\right) \tag{5.17}$$

Equation (5.17) states that water flows between river reach, $river$, and sub-aquifer, aqf, $X_{river,aqf,t}$, depend on river and sub-aquifer heads and hydraulic conductance of river reach, with $X_{river,aqf,t}$ being negative if sub-aquifer is discharging water to river reach.

5.2.2 Land use

For irrigated agriculture, land in production in each agricultural use node, (a subset of u), which derives water demand in that node, is defined in the following equations:

$$\sum_{j,k} L_{u,j,k,t} \leq Tland_{ut} \tag{5.18}$$

$$L_{u,per,k,t} \leq L_{u,per,k,t-1} \tag{5.19}$$

Equation (5.18) states that the sum over crops (j) and irrigation technologies (k) of irrigated land in production, $L_{u,j,k,t}$, at each agricultural use node in time, t, cannot exceed land availability, $Tland_{ut}$, in that use node and time period. Equation (5.19) states that irrigated land in production, $L_{u,per,k,t}$, of perennial crops, per (a subset of j), at each agricultural use node in time, t, cannot exceed perennial irrigated land for that use node in the previous time period, $t-1$. This constraint reflects the possible future loss of long-run capital investments in perennial crops if farmers decide to not irrigate those crops in the current time period.

5.2.3 Institutions and environment

Water agencies in arid and semiarid regions worldwide impose several institutional and environmental constraints on water use and management such allocations rules, minimum supply requirements, and minimum environmental flows. The reasons are the need to satisfy human water needs, to secure supply to downstream users, and to protect valuable aquatic ecosystems, among others.

In this paper, several institutional and environmental constraints are included depending on the climate and policy scenarios considered. A constraint on urban water supply is maintained in all scenarios in order to assure that a minimum amount of water, B_a^{min}, is delivered to urban application nodes, a, in each time period, t. This constraint is defined in the following form:

$$X_{a,t}^{urb} \geq B_a^{min} \tag{5.20}$$

5.2.4 Economics

Water has an economic value in all its competing uses, which is determined by the total willingness to pay of users benefiting from it. For agricultural use, the economic value of water is measured by the contribution of water to farmers' net benefits. For urban use, it is measured by the sum of the consumer and producer surplus.

Net benefits, $NB_{u,t}$, at each use node, u, in time, t, is defined as follows:

$$NB_{u,t} = TB_{u,t} - TC_{u,t} \tag{5.21}$$

where $TB_{u,t}$ and $TC_{u,t}$ are the total benefits and costs at each use node, u, in time, t, respectively.

For agricultural use nodes, total benefits, $TB_{u,t}^{ag}$, and total costs, $TC_{u,t}^{ag}$, in time, t, are defined by the following equations:

$$TB_{u,t}^{ag} = \sum_{j,k}\left(P_{u,j} \cdot Y_{u,j,k,t}\right) \cdot L_{u,j,k,t} \tag{5.22}$$

$$TC_{u,t}^{ag} = \sum_{j,k}\left(PC_{u,j,k,t} + WC_{u,j,k,t}\right) \cdot L_{u,j,k,t} \tag{5.23}$$

where parameters $P_{u,j}$ is crop prices; $Y_{u,j,k,t}$ is crop yields, and $PC_{u,j,k,t}$ is non-water production costs, and variable $L_{u,j,k,t}$ is crop area. Variable $WC_{u,j,k,t}$ is water costs which is defined as follows:

$$WC_{u,j,k,t} = PW_u \cdot \left(\sum_d B_{d,u} \cdot X_{d,j,k}\right) + \left(CP_{0,u} + CP_{1,u} \cdot PumpDepth_u\right) \cdot (\sum_p B_{p,u} \cdot X_{p,j,k}) \quad (5.24)$$

where parameters PW_u is surface water price, $CP_{0,u}$ is pumping cost not related to the level of the water table (investment, operation and maintenance of the well and pump equipment), and $CP_{1,u}$ is pumping cost related to the water table level or energy costs of lifting water from the water table to land surface. The variable $PumpDepth_u$ is the pumping depth, or the difference between the water table level (aquifer head) and land surface elevation. Variables $X_{d,j,k}$ and $X_{p,j,k}$ are the water applied to crops supplied with surface water and groundwater, respectively. Parameters $B_{d,u}$ and $B_{p,u}$ are vectors of coefficients that conform use nodes to diversion and pumping nodes, respectively.

For urban use nodes, (a subset of u), total benefits, $TB_{u,t}^{urb}$, and total costs, $TC_{u,t}^{urb}$, in time, t, are defined by the following equations:

$$TB_{u,t}^{urb} = \beta_{0,u} + \beta_{1,u} \cdot X_{a,t}^{urb} + \beta_{2,u} \cdot X_{a,t}^{urb\,2} \quad (5.25)$$

$$TC_{u,t}^{urb} = \delta_u \cdot X_{a,t}^{urb} \quad (5.26)$$

where equation (5.25) is the total benefits function with a quadratic specification, with $\beta_{0,u}$, $\beta_{1,u}$ and $\beta_{2,u}$ are the parameters for the constant, linear and quadratic terms, respectively. For urban use nodes, households utilize water first for high-value uses such as indoor uses for drinking, sanitation and cooking, so that urban benefits rise quickly for supplies allocated to these uses, starting from a position of no use. These high-value uses have few substitution possibilities, and therefore β_{1u} is expected to be large and positive. However, urban marginal benefits fall rapidly for other additional low-value uses, such as outdoor uses for garden irrigation and car washing. Then β_{2u} is expected to be large and negative. Equation (5.26) represents total urban supply costs, with δ_u being the per unit cost of water supplied.

5.2.5 Objective function

The model objective is maximizing the net present value of the economic net benefits over a planning horizon, subject to the basin's hydrological, land use, institutional, and environmental constraints. The model provides information on the optimal water flows and stocks, and cropping patterns under different climate and policy scenarios predefined by the modeler. The objective function takes the following form:

$$Max\ NPV = \sum_{u,t} \frac{NB_{u,t}}{(1+r)^t} \tag{5.27}$$

where NPV is the net present value, $NB_{u,t}$ are the net benefits, and r is the discount rate.

5.3 Model application

The modeling framework described previously in section 1 is applied to an arid and semiarid basin in Southeastern Spain, the Jucar basin. This basin is a good experimental field for an integrated basin scale analysis. One reason is that the Jucar is at present under severe stress with acute water scarcity and significant ecosystem degradation. Another reason is that the foreseeable climate change impacts are expected to exacerbate water scarcity problems in the basin. However, the modeling framework is designed to be adaptable for any basin elsewhere.

5.3.1 Study area: the Jucar basin

The Jucar basin is located in the regions of Valencia and Castilla La Mancha in Southeastern Spain. It extends over 22,300 km^2 and covers the area drained by the Jucar River and its tributaries, mainly the Magro and Cabriel Rivers. The basin is a complex system including 13 reservoirs and numerous competing uses with different priority rights, and with a complex relationship between surface and groundwater resources. The Jucar basin presents a ratio of 0.84 between total water demand and average renewable water resources. This value highlights the strong pressure on water resources in the basin (Momblanch et al., 2014).

Urban and industrial extractions are 270 Mm^3 to supply households, industries, and services in an area with more than one million inhabitants. This population is located mostly in the cities of Valencia, Sagunto and Albacete. Extractions for irrigated agriculture are nearly 1,400 Mm^3 to irrigate 190,000 ha. The main crops are rice, wheat, barley, corn, garlic, onion, grapes, and citrus. There are three major irrigation areas, the Eastern La Mancha irrigation area (EM) located in the upper Jucar; the traditional irrigation districts of Acequia Real del Jucar (ARJ), Escalona y Carcagente (ESC), and Ribera Baja (RB) located in the lower Jucar; and the irrigation area of the Canal Jucar-Turia (CJT) located in the bordering Turia Basin (CHJ, 2014).

The Jucar basin includes the Albufera wetland, which is one of the most important aquatic ecosystems in Europe. The Albufera is catalogued in the RAMSAR list, and it is a natural park and a special protected area for birds. It receives water mainly from the return flows of the ARJ and RB irrigation districts. Other flows originate from the neighboring Turia basin, and from the discharge of urban and industrial wastewaters in the adjacent municipalities (Sanchis, 2011).

Irrigation development during recent decades in the basin has been quite important for the local economy, and irrigated agriculture remains an important source of income and labor in the area. The expansion of irrigation has been driven especially by groundwater pumping from the EM aquifer, which is the largest aquifer system in Spain (Esteban and Albiac, 2012). However, the intensive groundwater pumping has caused a significant drop in the water table level reaching 80 m in some areas, and resulting in large storage depletion fluctuating around 2,500 Mm^3 at present. In addition, the EM aquifer is linked to the Jucar River stream, and it used to feed the river with about 200 Mm^3/year in the 1980s. Due to the depletion, aquifer discharges to the river have declined considerably over the past 30 years (Sanz et al., 2011). The consequence is that the lower Jucar is undergoing severe problems of low flows and water-quality degradation, with the riverbed in the middle Jucar being desiccated during recent droughts.

A major challenge for policymakers in the Jucar basin is the design of sustainable adaptation strategies to the upcoming effects of climate change, which is expected to reduce the freshwater supplies and increase the demand for water. Climate change projections for the end of the twenty-first century in the Jucar basin under a range of climatic and emission scenarios indicate a reduction of surface and groundwater availability between 11 and 46%, and an increase of evapotranspiration between 12 and 22% (CEDEX, 2010).

The hydro-economic modeling framework is applied to the Jucar basin in order to address adaptation to climate change. The analysis undertaken in this paper focuses on irrigation activities in the major irrigation districts (EM, CJT, ESC, ARJ and RB) and urban demand in the major cities (Albacete, Valencia, and Sagunto). Following the study by Sanz et al. (2011), the EM irrigation district is divided into three sub-areas of the aquifer, Northern Domain (NEM), Central Domain (CEM), and Southern Domain (SEM). In addition, the analysis includes the most important aquatic ecosystems in the Jucar basin: the Albufera wetland, the ecosystem linked to the Jucar River and its tributaries, and the groundwater-dependent ecosystems in the EM aquifer. Three proxy variables are used in order to quantify the environmental impacts of the climate and policy scenarios on these ecosystems: the inflows to the Albufera wetland, the outflows to the Mediterranean Sea, and the change in the EM aquifer storage. The model of the Jucar basin consists of 8 headwater inflow nodes, 21 river gauge nodes, 8 diversion nodes, 4 pumping nodes, 11 return flow nodes, 3 stream-aquifer interaction nodes, 1 environmental demand node, 3 reservoir release nodes, 3 reservoir stock nodes, and 3 aquifer stock nodes. Figure 5.1 presents the hydrological network of the basin, including the sources and uses of water.

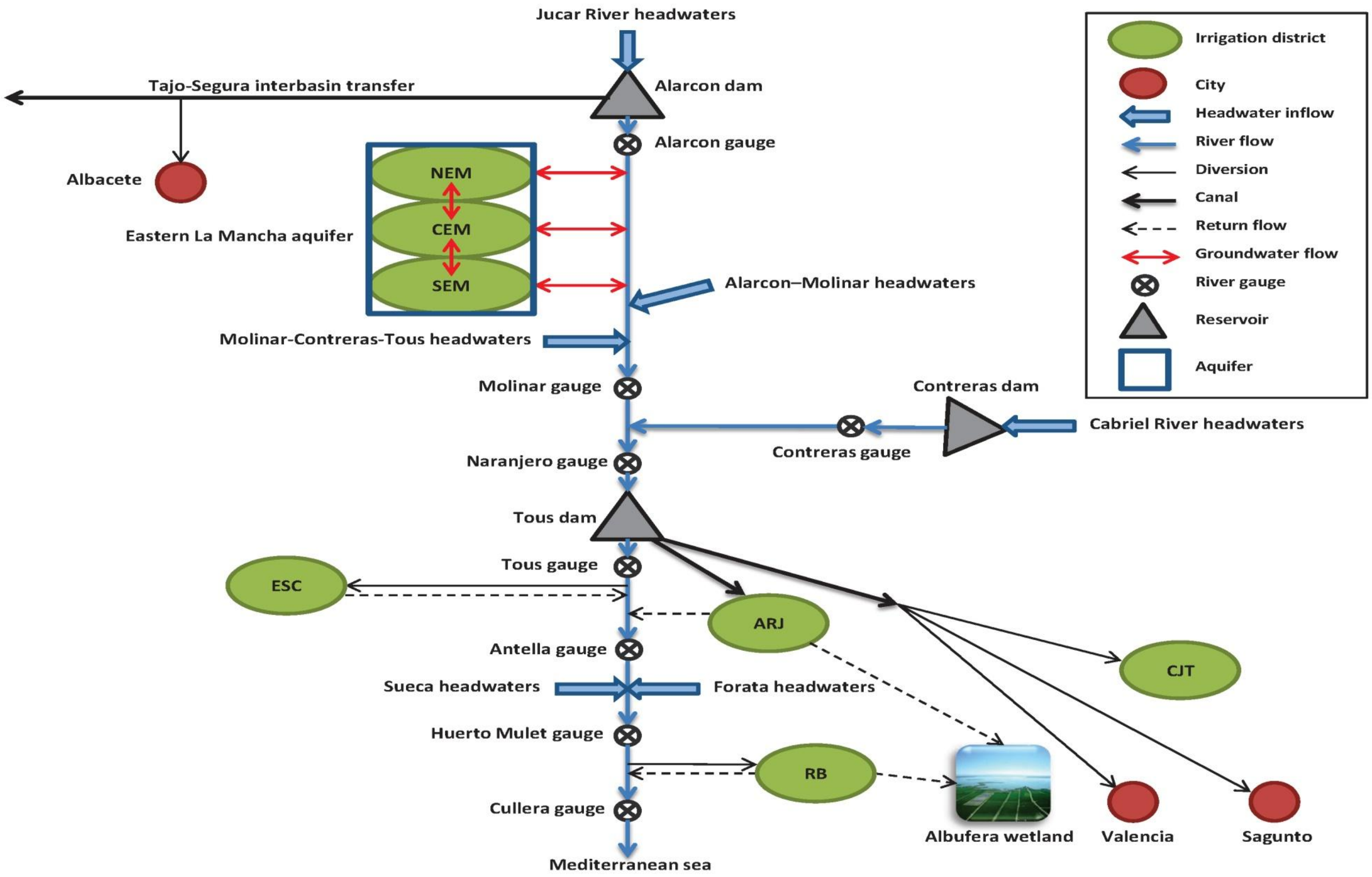
5.3.2 Data sources

Data on headwater inflows to the basin, gauged water flows, and reservoir inflows, releases and evaporation has been obtained from the reports of the Jucar basin authority and the Spanish Ministry of Agriculture and Environment (CHJ, 2014; MAGRAMA, 2014). Information on the parameters of the EM sub-aquifers including area, recharge, hydraulic conductance and specific yield has been taken from Sanz et al. (2011). Headwater inflows and aquifer recharge are stochastically represented in the model with means and variances of historical inflows and recharge, respectively.

For agricultural uses, detailed information on crop yields and prices, subsidies, crop water requirements, irrigation efficiencies, water and production costs, and land availability in each irrigation district have been collected from field surveys, expert consultation, statistical reports, and published documentation (INE, 2009; GV, 2009; GCLM, 2009; MARM, 2010). Irrigation water extractions by source of water in each district have been calculated using crop areas, water requirements, and location of irrigation technologies and their efficiencies. The crops included in the model are rice, wheat, barley, corn, other cereals, garlic, onion, other vegetables, citrus, grapes and other fruit trees. Irrigation technologies are flood, sprinkler and drip.

For urban uses, a linear demand function is specified to characterize the demand for water in each urban demand node. The linear demand function results in a quadratic benefit function similar to the one specified in equation (5.25). Parameter estimation requires three data items: the observed water price and quantity for a specific time period, and the price elasticity of water demand (Young, 2005). Information on urban water supply by source of water, population growth rate, water prices and costs has been obtained from the Jucar basin authority reports (CHJ, 2014). The price elasticity of

Figure 5.1. Network of the Jucar basin.



demand has been taken from Martinez-Espiñeira (2002) and Arbues and Barberan (2004).

The environmental benefits and damage costs for the most important aquatic ecosystems in the Jucar are estimated. For the Albufera wetland, an environmental benefit function of the wetland from chapter 1 of this thesis is used. For the Jucar River, a benefit function is specified as linear in the amounts of water in the mouth flowing to the Mediterranean Sea. We relied on valuation studies from the literature that estimate the values of the ecosystem services provided by rivers (Hatton et al., 2011, CSIRO, 2012, Banerjee et al., 2013). For groundwater-dependent ecosystems in the EM aquifer, a damage cost function is specified as linear in the volume of depletion following the study by Esteban and Albiac (2012).

Return flows to the Jucar River and to aquifers have been calculated as the fraction the applied water not used in crop evapotranspiration or in urban consumption. The information about the contribution of return flows to streamflow and aquifer recharge is taken from the reports of the Jucar basin authority (CHJ, 2014).

5.3.3 Model calibration

Integrated hydro-economic models typically require a careful calibration procedure before they can be used to assess sustainable water management policies. In this paper, both the hydrologic and the agricultural economic components of the Jucar model are calibrated. The calibration of the hydrologic component involves adjusting model parameters in order to reproduce the observed system states such as streamflows and aquifer heads under baseline conditions (Sophocleous et al., 2009). The agricultural economic component is calibrated using the Positive Mathematical Programming (PMP) in order to reproduce observed land and water use under baseline conditions, and to address the problem of overspecialization in agricultural production (Howitt, 1995). Both components are calibrated for the year 2009, which is a normal flow year.

The hydrological component is calibrated so that its predicted gauged flows match the observed flows at each river gauge, where measurement data are available (8 gauges in the Jucar). To achieve this, the model is constrained to reproduce observed gauged flows, and to deliver the observed water supply to irrigation districts and cities. The calibration procedure involves introducing new variables that represent unmeasured sources or uses of water, which allow balancing supply and demand at each node. These

Table 5.1. Climate change impacts in the Jucar basin compared to current climate.

Climate scenario	Mild	Severe
Temperature (ºC)	+3.8	+4.4
Rainfall (%)	-1	-24
Potential evapotranspiration (%)	+13	+22
Surface runoff (%)	-27	-46
Groundwater recharge (%)	-22	-45

Note: The mild climate change scenario is the outcome of the downscaled climatic model ECHAM4-FIC forced by the B2 emission scenario. The severe climate change scenario is the outcome of the downscaled climatic model HadCM3-SDSM forced by the A2 emission scenario. Both scenarios present projections for the period 2071-2100 compared to current climate conditions.

variables include all possible sources or uses of water in the basin that are not properly measured. Unmeasured sources include upstream headwater inflows, surface return flows, and aquifer discharge. Unmeasured uses include upstream demand nodes not included in the study, evapotranspiration of natural vegetation, evaporation from open water such as rivers and channels, and percolation. Additionally, the calibration procedure involves an adjustment of aquifer parameters such as hydraulic conductance, specific yield and recharge in order to reproduce the observed aquifer heads and the stream-aquifer interaction. The calibration procedure requires a fair amount of experimentation since the model have to be calibrated node by node from upstream to downstream. Once the model calibration is satisfactory, all unmeasured sources and uses have to be held constant. Then any changes brought about by new policy intervention scenarios will not change these unmeasured levels.

The agricultural economic component is calibrated using a variant of PMP developed by Dagnino and Ward (2012), in which parameters are estimated for a linear crop yield function. This function represents a decreasing crop yield when additional land is assigned to crop production, based on the principle of Ricardian rent. For each crop and irrigation technology, the first lands brought into production have the highest yields, after which yields fall off as less-suitable lands enter production. The parameters of the linear yield function for each crop and irrigation technology are given in tables B1 and B2 in the appendix.

5.3.4 Climate change and policy scenarios

The modeling framework is used to analyze climate change impacts and adaptation possibilities under various climate and policy scenarios in the Jucar basin. Two climate change scenarios are considered: mild and severe. These scenarios cover climate change impacts on potential evapotransipration, surface runoff, and groundwater recharge as shown in table 5.1. Impact estimates are taken from climate change projections for the Jucar basin by CEDEX (2010), which downscales to basin level the results of various global circulation models and emission scenarios.

The model is used to assess the outcomes of two policy alternatives under the climate change scenarios defined above. The two policy alternatives are defined as follows:

***Unsustainable management policy*:** This policy promotes a high use of water which is above renewable water availability. The policy is implemented in the model by placing no requirements on terminal reservoir or aquifer stocks, or on yearly streamflows. Reservoirs and aquifers can be run down as low as desired up to the last time period with no regard for future water uses or for environmental damages caused by water resources depletion. The cost that groundwater users confront when pumping aquifers unsustainably is the increased pumping costs incurred by lowering the aquifer heads. Under unsustainable management, competing users ignore the common pool nature of groundwater creating the water extraction externality, where extractions by one user reduce the water stock available for others. Because every user believes that competitors will not conserve water for future use, there is no incentive to protect the water stock. Pumping by users takes place as long as the economic value of the marginal product of pumped water exceeds the marginal pumping cost. Beyond these marginal costs, there are no incentives to conserve water for the future or account for other environmental externalities related to groundwater depletion (Esteban and Albiac, 2012).

In recent decades, aquifer systems have been suffering substantial pressures in arid and semi-arid regions, with extraction rates well above recharge (Richey et al. 2015). Significant negative impacts are already occurring in many basins worldwide, because the degradation of water bodies limits economic activities and endangers ecosystems (UNEP 2003; WWAP 2006). In addition, individual agents are unable to capture the future value of stock resources. Therefore, both surface water stored in reservoirs and groundwater resources in the absence of adequate regulation are misallocated and used more intensively than what is socially desirable (Esteban and Albiac, 2012).

***Sustainable management policy*:** This policy promotes the protection of water resources, accounting for long-term and environmental benefits. The sustainable management of water resources requires a reform of the water institutions and policies used at present that have failed to align private short-term goals with societal long-term goals (Guerry et al., 2015). For the purpose of this paper, sustainable water management is defined as the water extractions that do not exceed the natural replenishment rate and maintain minimum environmental flow thresholds. This policy is implemented by requiring that all aquifers and reservoirs in the basin return to their starting levels by the end of the planning period, and that annual streamflows are greater than the minimum flow thresholds set for the Jucar River.

These two policy alternatives do not necessarily replicate the current water management approach in the Jucar basin, but they provide a range of the possible future climate change impacts under different water management policy choices.

5.3.5 Solving the model

The model is formulated as a dynamic nonlinear problem that maximizes the Jucar basin's net present value for a 20 years' time period. The GAMS package has been used for model development and scenario simulation (Brooke et al., 1988). The dimensions of the model are 391,317 equations, 421,764 variables and 1,039,011 nonzero elements. The model is solved using the CONOPT algorithm within GAMS, which is designed to solve large-scale nonlinear optimization models.

5.4 Results and discussion

The results for the climate change and policy scenarios are compared to those of the current situation or baseline in terms of hydrologic, land use and economic outcomes. Results are presented by demand node, sector and basin location. The tables show average values for the analyzed planning period.

5.4.1 Baseline scenario

Table 5.2 shows the outcomes of the baseline scenario. The hydrologic outcomes of this scenario indicate that total water demand is 799 Mm^3 per year, divided between 690 Mm^3 for agricultural demand (86%) and 110 Mm^3 for urban demand (14%).[6] The surface water diversions are 483 Mm^3 covering the agricultural and urban demand, especially in the lower Jucar region of Valencia. These surface water extractions do not affect reservoir storage, which increases by 10 Mm^3 per year. Groundwater extractions

[6] About 260 Mm^3/year of water extractions by numerous small demand nodes are not included in the model.

Table 5.2. Hydrologic, land use and economic outcomes of the baseline scenario.

Region/basin location	**Castilla La Mancha/Upstream**				**Valencia/Downstream**						**Basin**			
Sector	**Agriculture**			**Urban**	**Agriculture**				**Urban**		**Agriculture**	**Urban**	**Environment**	**Total**
Demand nodes	**NEM**	**CEM**	**SEM**	**Albacete**	**CJT**	**ESC**	**ARJ**	**RB**	**Valencia**	**Sagunto**				
Hydrologic outcomes (Mm³/year)														
Headwater inflows														1355.5
Aquifer recharge														323.1
Water demand	16.3	185.9	58.4	15.3	112.5	18.8	104.5	193.6	87.9	6.2	689.9	109.5		799.4
Surface water diversion	0.0	0.0	0.0	15.3	56.5	18.8	104.5	193.6	87.9	6.2	373.4	109.5		482.9
Groundwater pumping	16.3	185.9	58.4	0.0	56.0	0.0	0.0	0.0	0.0	0.0	316.5	0.0		316.5
Storage change (storage depletion if <0)														
Reservoirs														9.9
Aquifers													-39.3	-39.3
Aquifer-river discharge (river gains if >0)														45.9
Outflow to Mediterranean Sea													416.6	416.6
Inflows to Albufera wetland													88.6	88.6
Land use outcomes														
Irrigated area (1000 ha/year)*	6.8	45.9	17.1		19.2	3.4	15.3	15.3			123.0			123.0
Cereals	2.9	27.3	11.1		0.5	0.0	3.1	8.6			53.5			53.5
Vegetables	0.5	10.5	3.5		0.7	0.0	0.6	0.2			16.0			16.0
Fruit trees	3.4	8.1	2.6		18.0	3.4	11.6	6.4			53.5			53.5
Irrigation technology distribution (%)														
Flood	1.8	4.0	4.3		23.9	38.7	50.8	69.1			21.9			21.9
Sprinkler	42.6	59.5	64.6		0.1	0.0	0.5	0.1			33.7			33.7
Drip	55.6	36.4	31.1		76.0	61.3	48.7	30.9			44.4			44.4
Economic outcomes														
Gross benefits (million €/year)	11.1	96.8	32.5	75.1	94.4	16.8	66.8	49.2	430.9	30.6	367.4	536.6	205.6	1109.6
Production costs (million €/year)	7.1	60.0	20.3	19.8	71.0	13.4	51.5	38.1	113.4	8.1	261.4	141.3	1.3†	404.0
Net benefits (million €/year)	4.0	36.7	12.2	55.3	23.4	3.4	15.3	11.1	317.5	22.5	106.0	395.3	204.3	705.6
Marginal value of irrigation water (€/m³)	0.10	0.11	0.09		0.08	0.03	0.03	0.01			0.06			
Urban water price (€/m³)				1.29					1.29	1.29		1.29		

* Crops are aggregated into three representative groups: cereals: rice, wheat, barley, corn, other cereals; vegetables: garlic, onion, other vegetables and Fruit trees: citrus, grapes and other fruit trees.

† For the environment, production costs are equivalent to damage costs.

are 317 Mm^3 and they are the major water sources for the irrigation districts located in the region of Castilla La Mancha in the upper Jucar (NEM, CEM and SEM).

Results show that under the current policy setting and climate conditions, groundwater pumping results in aquifer depletion of about 39 Mm^3 per year. The consequence is that aquifer discharge to the river is no more than 46 Mm^3 per year, which is very low compared to the historical discharges of 250 Mm^3 before the largest pumping extractions took place in the 1999's (Sanz et al., 2011). The annual water outflow to the Mediterranean Sea is 417 Mm^3, well above the annual environmental flow threshold required to achieve a good ecological status of the Jucar River (63 Mm^3 or 2 m^3/s). The Albufera wetland receives about 89 Mm^3 per year from irrigation return flows, which meets the wetland water requirements in order to achieve a good ecological status (CHJ, 2014).

The land use outcomes show that the irrigated area amounts to 123,000 ha per year, of which 53,500 ha are cereals, 16,000 ha are vegetables, and 53,500 ha are fruit trees. A considerable irrigated area is grown under high-efficient irrigation technologies (34% sprinkler and 44% drip), especially in the upper Jucar. About one fifth of the irrigated area is grown under low-efficient flood irrigation technology, especially in the lower Jucar.

The economic outcomes indicate that the basin net benefits are 706 million €. Agriculture, which is the major water user in the basin, produces only 15% of net benefits. Environmental uses generate 29% of net benefits. The major share of net benefits accrues to urban uses, about 56% of the total. This large share of benefits calculated for urban uses occurs because of the low price elasticity of demand for urban uses and its associated high consumer surplus. The economic outcomes reflect the intense competition for water between agriculture, urban and environmental uses.

The last two rows in table 5.2 show the economic value of an additional cubic meter of water (or shadow price) for farmers and households from water reallocation or supply increases. These shadow prices provide important information to policymakers on the willingness to pay for water by users, they could guide allocation decision, and they could indicate whether investments in developing alternative sources of water such as desalination and water conservation are required or not. Results show that the shadow price of water is very high for urban use compared to agricultural use. These results justify the fact that agriculture usually faces the main adjustment to water scarcity. The marginal values of irrigation water are higher in the upper Jucar, where groundwater resources are intensively used, compared to those in the lower Jucar based mostly on surface water.

5.4.2 Mild climate change scenario

Tables 5.3 and 5.4 show the outcomes of the mild climate change scenario under the two alternative water management policies, unsustainable and sustainable management. Under this climate scenario, headwater inflows are reduced by 30%. Aquifer recharge is reduced by 21 and 27% under the unsustainable and sustainable management policies, respectively. Total water demand is reduced by 5 and 19% under the unsustainable and sustainable policies, respectively.

The economic outcomes of this scenario indicate that the mild climate change scenario reduces net benefits between 85 and 91 million € per year (up to 13%) compared to the baseline scenario. However, contrary to expectations the sustainable policy achieves higher net benefits compared to the unsustainable policy because the environmental net benefit gains (+8%) outweigh the agricultural net benefit losses (-4%) in the sustainable policy. Urban net benefits for both policies remain almost unchanged under this climate change scenario compared to the baseline because of the very small reduction in urban water supply. Urban water prices rise slightly by 1 and 2% under the unsustainable and sustainable policies, respectively.

The major impact of climate change falls on agriculture and the environment, which sustain the costs of adaptation. The reason is the large cutbacks in agriculture allocations coupled with depleted water stocks and river flows. Agriculture gets more benefits under the unsustainable policy because this policy increases both surface and groundwater extractions, drawing from the water stocks in reservoirs and aquifers, and river flows. Under mild climate change and the unsustainable policy, reservoir depletion is 10 Mm^3 per year, and aquifer depletion is 65 Mm^3 per year.

The sustainable policy, which avoids water stocks depletion and assures minimum river flows achieves higher environmental net benefits (about 8%) compared to the unsustainable policy. The aquifer discharge to the river increases under the sustainable policy compared to the unsustainable policy and the baseline scenario. This increase in aquifer discharges to the river enhances river flows available for water users

Table 5.3. Hydrologic, land use and economic outcomes of the mild climate change scenario and unsustainable management policy.

Region/basin location	**Castilla La Mancha/Upstream**				**Valencia/Downstream**						**Basin**			
Sector	**Agriculture**			**Urban**	**Agriculture**				**Urban**		**Agriculture**	**Urban**	**Environment**	**Total**
Demand nodes	**NEM**	**CEM**	**SEM**	**Albacete**	**CJT**	**ESC**	**ARJ**	**RB**	**Valencia**	**Sagunto**				
Hydrologic outcomes (Mm³/year)														
Headwater inflows														949.0
Aquifer recharge														255.2
Water demand	16.4	161.4	53.6	15.3	108.9	18.0	82.8	210.8	87.7	6.2	651.8	109.3		761.0
Surface water diversion	0.0	0.0	0.0	15.3	65.2	18.0	82.8	210.8	87.7	6.2	376.7	109.3		486.0
Groundwater pumping	16.4	161.4	53.6	0.0	43.7	0.0	0.0	0.0	0.0	0.0	275.1	0.0		275.1
Storage change (storage depletion if <0)														
Reservoirs														-10.1
Aquifers													-64.7	-64.7
Aquifer-river discharge (river gains if >0)														44.9
Outflow to Mediterranean Sea													98.1	98.1
Inflows to Albufera wetland													83.6	83.6
Land use outcomes														
Irrigated area (1000 ha/year)	6.2	36.3	14.3		16.7	2.9	11.8	14.8			103.0			103.0
Cereals	2.5	19.2	8.8		0.2	0.0	1.3	8.2			40.1			40.1
Vegetables	0.5	9.4	3.1		0.6	0.0	0.5	0.2			14.4			14.4
Fruit trees	3.3	7.7	2.4		15.8	2.9	10.0	6.4			48.5			48.5
Irrigation technology distribution (%)														
Flood	1.4	2.9	3.1		22.1	37.2	43.7	68.4			21.0			21.0
Sprinkler	39.3	52.8	61.5		0.1	0.0	0.3	0.1			29.6			29.6
Drip	59.2	44.3	35.4		77.9	62.8	56.0	31.5			49.4			49.4
Economic outcomes														
Gross benefits (million €/year)	10.5	87.1	29.7	75.1	86.6	15.1	58.9	48.5	430.6	30.6	336.3	536.3	122.7	995.3
Production costs (million €/year)	6.9	53.8	18.6	19.8	64.5	11.8	44.4	37.7	113.1	8.0	237.6	141.0	2.1	380.7
Net benefits (million €/year)	3.7	33.3	11.1	55.3	22.1	3.3	14.5	10.8	317.5	22.5	98.7	395.3	120.6	614.6
Marginal value of irrigation water (€/m³)	0.11	0.11	0.09		0.09	0.03	0.04	0.01			0.07			
Urban water price (€/m³)				1.29					1.31	1.31		1.30		

Note: see note to table 5.2.

Table 5.4. Hydrologic, land use and economic outcomes of the mild climate change scenario and sustainable management policy.

Region/basin location	**Castilla La Mancha/Upstream**				**Valencia/Downstream**						**Basin**			
Sector	**Agriculture**			**Urban**	**Agriculture**				**Urban**		**Agriculture**	**Urban**	**Environment**	**Total**
Demand nodes	**NEM**	**CEM**	**SEM**	**Albacete**	**CJT**	**ESC**	**ARJ**	**RB**	**Valencia**	**Sagunto**				
Hydrologic outcomes (Mm³/year)														
Headwater inflows														949.0
Aquifer recharge														236.9
Water demand	6.3	104.3	31.3	15.3	100.5	16.1	63.8	212.9	87.6	6.2	535.2	109.1		644.3
Surface water diversion	0.0	0.0	0.0	15.3	56.8	16.1	63.8	212.9	87.6	6.2	349.6	109.1		458.7
Groundwater pumping	6.3	104.3	31.3	0.0	43.7	0.0	0.0	0.0	0.0	0.0	185.7	0.0		185.7
Storage change (storage depletion if <0)														
Reservoirs														0.0
Aquifers													0.0	0.0
Aquifer-river discharge (river gains if >0)														51.2
Outflow to Mediterranean Sea													148.9	148.9
Inflows to Albufera wetland													76.3	76.3
Land use outcomes														
Irrigated area (1000 ha/year)	3.0	24.4	8.6		15.5	2.6	9.9	15.0			79.0			79.0
Cereals	0.3	9.1	4.2		0.1	0.0	0.3	8.3			22.3			22.3
Vegetables	0.3	8.1	2.5		0.6	0.0	0.5	0.2			12.2			12.2
Fruit trees	2.4	7.2	1.9		14.8	2.6	9.1	6.4			44.5			44.5
Irrigation technology distribution (%)														
Flood	0.0	0.8	0.5		21.0	36.1	37.8	68.6			23.3			23.3
Sprinkler	9.8	37.1	48.9		0.1	0.0	0.1	0.1			17.2			17.2
Drip	90.2	62.0	50.6		78.9	63.9	62.0	31.4			59.4			59.4
Economic outcomes														
Gross benefits (million €/year)	6.9	73.1	23.2	75.1	82.2	13.9	53.1	48.7	430.4	30.6	301.0	536.1	130.3	967.4
Production costs (million €/year)	3.9	40.7	12.7	19.7	60.4	10.7	39.5	37.9	113.0	8.0	205.7	140.8	0.0	346.5
Net benefits (million €/year)	3.0	32.4	10.5	55.3	21.8	3.2	13.6	10.8	317.5	22.5	95.3	395.3	130.3	620.9
Marginal value of irrigation water (€/m³)	0.14	0.12	0.10		0.09	0.03	0.05	0.01			0.08			
Urban water price (€/m³)				1.30					1.31	1.31		1.31		

Note: see note to table 5.2.

downstream, and therefore puts less pressure on the water stocks in reservoirs that can be maintained.

Compared to the baseline scenario, the water flowing to the Mediterranean Sea decreases considerably under climate change for the two policies (up to 76%), but this water flow is higher under the sustainable than under unsustainable policies. Nevertheless, outflows to sea under both policies comply with the small minimum environmental flow threshold. The inflows to the Albufera wetland decrease under the mild climate change compared to the baseline scenario. The wetland receives larger inflows under the unsustainable than the sustainable policy. The reason is that the Albufera wetland is fed by irrigation return flows in the lower Jucar, which are reduced under the sustainable policy as a result of the decline in water extractions.

5.4.3 Severe climate change scenario

Tables 5.5 and 5.6 show the outcomes from severe climate change under the two alternative policies. Under this scenario, headwater inflows are reduced by 48%. Aquifer recharge is reduced by 43 and 52% under the unsustainable and sustainable policies, respectively. Water demand falls by 19 and 43% under the unsustainable and sustainable policies, respectively.

The severe climate change scenario reduces basin net benefits between 133 and 147 million € per year (up to 21%) compared to the baseline scenario. The sustainable policy results in larger benefit losses compared to the unsustainable policy because the gains in environmental benefits (+15%) do not cover the agricultural benefit losses (-30%). Urban benefits for both policies remain almost unchanged because of the small reduction in urban water supply. Urban water prices rise slightly by 3 and 5% under the unsustainable and sustainable policies, respectively.

The impacts of severe climate change on agriculture are considerable with benefits dropping between 15 and 40%, compared to the baseline. The cost of achieving sustainability under severe climate change is supported by agriculture with benefits falling 30% in comparison to the unsustainable policy. Without sustainability requirements, the depletion levels in reservoirs and aquifers are 10 and 92 Mm^3 per year, respectively. The marginal value of irrigation water increases under severe climate change scenario, and it is even higher for the sustainable policy where less water is available for irrigation.

Table 5.5. Hydrologic, land use and economic outcomes of the severe climate change scenario and unsustainable management policy.

Region/basin location	**Castilla La Mancha/Upstream**				**Valencia/Downstream**						**Basin**			
Sector	**Agriculture**			**Urban**	**Agriculture**				**Urban**		**Agriculture**	**Urban**	**Environment**	**Total**
Demand nodes	**NEM**	**CEM**	**SEM**	**Albacete**	**CJT**	**ESC**	**ARJ**	**RB**	**Valencia**	**Sagunto**				
Hydrologic outcomes (Mm^3/year)														
Headwater inflows														706.5
Aquifer recharge														184.9
Water demand	15.9	137.7	48.6	15.3	86.7	12.5	49.2	185.4	87.3	6.2	536.1	108.8		644.9
Surface water diversion	0.0	0.0	0.0	15.3	55.9	12.5	49.2	185.4	87.3	6.2	303.0	108.8		411.8
Groundwater pumping	15.9	137.7	48.6	0.0	30.8	0.0	0.0	0.0	0.0	0.0	233.1	0.0		233.1
Storage change (storage depletion if <0)														
Reservoirs														-10.1
Aquifers													-92.4	-92.4
Aquifer-river discharge (river gains if >0)														44.2
Outflow to Mediterranean Sea													31.5	31.5
Inflows to Albufera wetland													63.5	63.5
Land use outcomes														
Irrigated area (1000 ha/year)	5.8	29.7	12.3		12.6	1.9	7.4	12.7			82.3			82.3
Cereals	2.1	13.5	7.2		0.0	0.0	0.0	6.4			29.2			29.2
Vegetables	0.4	8.6	2.9		0.5	0.0	0.4	0.2			13.0			13.0
Fruit trees	3.3	7.5	2.2		12.0	1.9	7.0	6.1			40.0			40.0
Irrigation system distribution (%)														
Flood	1.1	1.6	1.9		18.8	32.1	32.5	64.4			17.4			17.4
Sprinkler	36.2	45.6	58.4		0.0	0.0	0.0	0.1			27.7			27.7
Drip	62.7	52.8	39.6		81.2	67.9	67.5	35.5			54.8			54.8
Economic outcomes														
Gross benefits (million €/year)	10.1	79.4	27.4	75.1	73.8	11.9	45.6	44.6	430.1	30.5	292.7	535.7	89.7	918.1
Production costs (million €/year)	6.6	48.4	17.2	19.7	53.5	9.0	33.2	34.3	112.6	8.0	202.1	140.3	3.0	345.4
Net benefits (million €/year)	3.4	31.0	10.2	55.3	20.3	2.9	12.3	10.3	317.5	22.5	90.6	395.3	86.7	572.6
Marginal value of irrigation water (€/m^3)	0.11	0.12	0.10		0.10	0.04	0.06	0.01			0.08			
Urban water price (€/m^3)				1.30					1.34	1.34		1.33		

Note: see note to table 5.2.

Table 5.6. Hydrologic, land use and economic outcomes of the severe climate change scenario and sustainable management policy.

Region/basin location	**Castilla La Mancha/Upstream**				**Valencia/Downstream**						**Basin**			
Sector	**Agriculture**			**Urban**	**Agriculture**				**Urban**		**Agriculture**	**Urban**	**Environment**	**Total**
Demand nodes	**NEM**	**CEM**	**SEM**	**Albacete**	**CJT**	**ESC**	**ARJ**	**RB**	**Valencia**	**Sagunto**				
Hydrologic outcomes (Mm^3/year)														
Headwater inflows														706.5
Aquifer recharge														155.6
Water demand	0.0	12.2	18.3	15.3	74.5	9.2	38.3	193.1	87.1	6.2	345.6	108.5		454.1
Surface water diversion	0.0	0.0	0.0	15.3	43.7	9.2	38.3	193.1	87.1	6.2	284.2	108.5		392.7
Groundwater pumping	0.0	12.2	18.3	0.0	30.8	0.0	0.0	0.0	0.0	0.0	61.4	0.0		61.4
Storage change (storage depletion if <0)														
Reservoirs														0.0
Aquifers													36.2	36.2
Aquifer-river discharge (river gains if >0)														58.0
Outflow to Mediterranean Sea													73.9	73.9
Inflows to Albufera wetland													61.3	61.3
Land use outcomes														
Irrigated area (1000 ha/year)	0.0	2.5	4.4		10.9	1.4	5.8	13.1			38.1			38.1
Cereals	0.0	0.0	1.3		0.0	0.0	0.0	6.7			8.1			8.1
Vegetables	0.0	2.5	2.0		0.5	0.0	0.3	0.2			5.5			5.5
Fruit trees	0.0	0.0	1.0		10.4	1.4	5.6	6.2			24.6			24.6
Irrigation system distribution (%)														
Flood	0.0	0.0	0.0		17.2	27.2	28.0	65.3			32.7			32.7
Sprinkler	0.0	1.6	30.0		0.0	0.0	0.0	0.1			3.6			3.6
Drip	100.0	98.4	70.0		82.8	72.8	72.0	34.7			63.8			63.8
Economic outcomes														
Gross benefits (million €/year)	0.5	20.8	18.1	75.0	66.2	9.7	38.5	45.3	429.8	30.5	199.3	535.3	99.6	834.2
Production costs (million €/year)	0.4	9.9	9.3	19.7	46.9	7.2	27.4	34.9	112.3	8.0	135.8	140.0	0.0	275.8
Net benefits (million €/year)	0.2	11.0	8.9	55.3	19.3	2.6	11.2	10.4	317.4	22.5	63.5	395.3	99.6	558.4
Marginal value of irrigation water (€/m^3)	0.24	0.17	0.11		0.11	0.04	0.06	0.01			0.11			
Urban water price (€/m^3)				1.32					1.36	1.36		1.35		

Note: see note to table 5.2.

Policymakers in arid and semiarid regions worldwide are constantly searching for policies leading to the sustainable use of water resources, mostly linked to reductions in overall basin extractions. The cost of such policies are given in terms of benefits losses (or gains) sustained by the groups of stakeholders. For policy success, the costs of these policies should be acceptable to stakeholders, eventually through compensation of losers. Otherwise, stakeholders will oppose any sustainable measure, leading to policy failure.

Table 5.6 shows how to meet sustainable outcomes under severe climate change in the Jucar basin. The objective is finding water allocations which have reasonable policy costs, measured by reduction in the present value of the stream of benefits along the planning horizon. Results indicate that the best way to achieve that is by substantially reducing groundwater pumping in the upper Jucar, and increasing the surface water available to downstream users.

Pumping in the upper Jucar under the sustainable policy is reduced by 85% compared to the unsustainable policy, down to levels well below aquifer recharge. This occurs because the aquifer head rises when pumping is less than recharge, allowing larger discharges from the aquifer to the river. Therefore, higher amounts of water are available in the river satisfying environmental flows requirements, and at the same time providing water to downstream surface water users that cannot get water by depleting reservoirs. Benefits of irrigation districts in the upper Jucar under the sustainable policy fall by 55% compared to the unsustainable policy. However, the benefits of irrigation districts in the lower Jucar are slightly reduced under the sustainable policy compared to the unsustainable policy. Water flowing to the sea decreases substantially under severe climate change, between 82 and 92% compared to the baseline scenario. Under the unsustainable policy, outflows are below the minimum environmental flow requirement, while the sustainable policy satisfies this requirement. Inflows to the Albufera wetland are also reduced under severe climate change compared to the baseline scenario. Inflows to the wetland are lower under the sustainable policy compared to the unsustainable policy, because the smaller water extractions reduce also the return flow feeding the wetland.

Table 5.7. The present value of benefits by climate and policy scenario (million €).

Policy scenario	Climate scenario	Municipal	Agriculture	Environment	Total private benefits	Total social benefits
Base	Normal	5101.1	1389.6	2653.6	6490.7	9144.3
Unsustainable policy	Mild	5101.1	1285.7	1569.1	6386.9	7956.0
	Severe	5100.9	1162.1	1125.1	6263.0	7388.0
Sustainable policy	Mild	5101.1	1236.9	1714.6	6338.0	8052.6
	Severe	5100.7	792.2	1326.0	5892.9	7219.0

Note: Total private benefits are the sum of municipal and agricultural benefits, while total social benefits are the sum of private and environmental benefits.

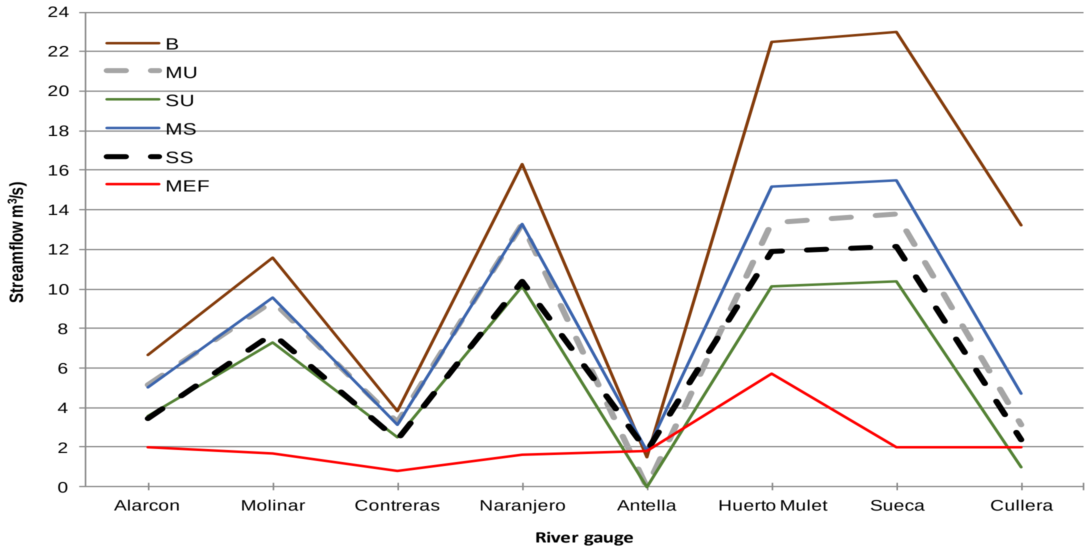
5.4.4 Tradeoffs among policies

The comparison between climate and policy scenarios shows the environmental and economic tradeoffs among policy choices. This information could be useful for the design of sustainable climate change adaptation policies at basin scale. Table 5.7 displays the present value of benefits for the climate and policy scenarios. Results indicate that climate change will have negative effects on the basin social benefits for the considered climate and policy scenarios. Benefits decline between 12 and 21% under climate change. However, the losses of private benefits are less than 10%. The impacts vary by group of users, with urban uses not very affected, and agricultural and environmental users bearing quite large damages.

Results show that the impacts of climate change depend on policy choices. The adaptation of stakeholders can be economically efficient, but this does not guarantee sustainable outcomes. In absence of regulations protecting the natural environment and the stock resources, water users will strategically deplete reservoirs, aquifers and river flows to better engage the impacts of climate change. But this involves serious damages to water-dependent ecosystems and also threatens future human activities. Conversely, the inclusion of sustainability objectives within the adaptation policies reduces the climate change impacts on the environment, but leads to very costly impacts on current economic activities.

For agriculture, there is a substantial gap between the benefits obtained under severe climate change and sustainable policy, and all the other scenarios. This negative impact of combining severe climate change with sustainable policy is too detrimental to farmers, and the costs of the policy become prohibitive. Therefore, additional policy instruments are needed to compensate farmers for their large benefit losses such as providing them with payments for the water released to support ecosystem services.

Figure 5.2. Streamflow in different river gauges under the climate and policy scenarios.

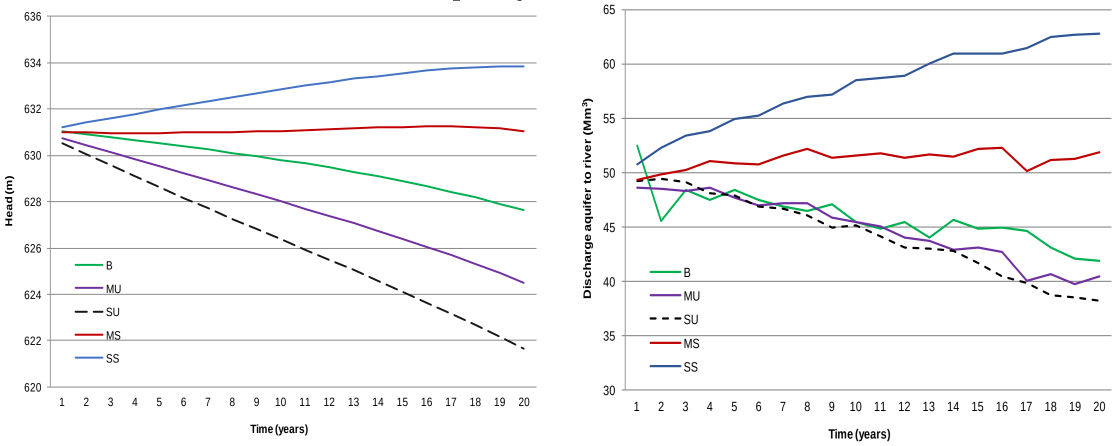


Note: B=Baseline scenario; MU=Mild climate change and unsustainable policy; SU=Severe climate change and unsustainable policy; MS=Mild climate change and sustainable policy; SS=Severe climate change and sustainable policy; MEF=Minimum environmental flows.

For environmental uses, the sustainable management policy reduces the negative impacts of climate change by increasing river flows and avoiding the depletion of aquifers and reservoirs. However, the Albufera wetland does not benefit from the sustainable policy because the Albufera depends on the irrigation return flows which diminish under the sustainable policy. A possible solution to recover water for the Albufera wetland could be the direct allocation of some river flow gains to the wetland.

Figure 5.2 shows the average river flow over the 20 year planning horizon in different river gauges under alternative climate and policy scenarios. River flows, which are the main drivers to maintain the river's good ecological status, decline under all climate change and policy scenarios compared to the baseline. The decline is especially remarkable in the downstream gauges (from Antella to Cullera) where the basin's major surface water users are located. However, river flow is higher in all gauges under the sustainable policy compared to the unsustainable policy. Non-compliance with the small environmental flow requirements occurs only in the Antella and Cullera gauges. Non-compliance in Antella occurs under mild or severe climate change for the unsustainable policy. Non-compliance in Cullera occurs only under severe climate change for the unsustainable policy.

Figure 5.3. Aquifer head (right) and discharge to the river (left) under the climate and policy scenarios.



Note: See note to figure 5.2. Aquifer head and discharge in each year are average values for the three sub-aquifers.

Figure 5.3 shows the paths of the aquifer head and discharge from the aquifer to the river along the 20 years planning horizon for the climate and policy scenarios. Results from tables 5.2, 5.3 and 5.5 indicate that without sustainability requirements, groundwater pumping in the upper Jucar is very high compared to aquifer recharge. Pumping extractions amounts to 98% of recharge for the baseline scenario, 109% of recharge for the mild climate change, and 131% of recharge for the severe climate change. The consequence of the unsustainable policy is a steady drop in both the water table level and the aquifer discharges to the river. Under the sustainable policy, the water table recovers and discharges from the aquifer to the river increase, because farmers reduce pumping down to 74 and 25% of recharge for the mild and severe climate change, respectively.

5.5 Conclusions

River basins in arid and semiarid regions worldwide face important water scarcity challenges, which will be aggravated by climate change in the coming decades. Policymakers in these basins have to make difficult decisions on water management and policies that involve complex environmental and economic tradeoffs. Solving these challenges requires better analytical tools to advance more sustainable management and policy options. A key task is the integration of the complex interrelationships between hydrological, economic, institutional and environmental components in basins.

Hydro-economic modeling is an emerging tool for implementing comprehensive basin scale analysis that could inform the design of sustainable water management policies. However, hydro-economic models have to be capable to adequately reproduce the physical behavior of the basin, with a realistic representation of the different water sources and uses, including the interaction between surface water and groundwater, as well as the value of the alternative water allocations. This paper has addressed that challenge by developing an integrated hydro-economic model which is applied to the assessment of climate change scenarios and policy choices in the Jucar basin of Spain. The contribution of this paper to previous hydro-economic modeling efforts stems from the improvement of the river basin dynamics. A groundwater flow framework similar to the MODFLOW groundwater model is added to the standard hydro-economic formulation of basins. This improved methodological approach is capable of simulating the spatial and temporal heterogeneity of real-world aquifers, and most important the linkages between aquifer systems and river flows.

Results of applying the modeling framework to the Jucar basin demonstrate the model capabilities to assess the climate scenarios and policy choices, and also its potential for integrating the multiple dimensions of water resources. The results of the climate change and policy scenarios provide information on the spatio-temporal impacts of climate change on hydrology, land use and economic values. Results illustrate how adaptation to climate change could be strategically undertaken at basin scale, showing also the economic and environmental tradeoffs among the water policy choices. Such information, which could be provided only by hydro-economic models, is crucial to assist policymakers in arid and semiarid basins in the design and implementation of sustainable water management policies.

5.6 References

Arbues, F., Barberan, R., 2004. Price impact on urban residential water demand: a dynamic panel data approach. *Water Resour. Res.* 40, W11402, doi:10.1029/2004WR003092.

Banerjee, O., Bark, R., Connor, J., Crossman, N., 2013. An ecosystem services approach to estimating economic losses associated with drought. *Ecol. Econ.* 91, 19-27.

Brooke, A., Kendrick, D., Meeraus, A., 1988. GAMS a User's Guide. The Scientific Press.

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

Brozovic, N., Sunding, D., Zilberman, D., 2010. On the spatial nature of the groundwater pumping externality. *Resour. Energy Econ.* 32, 154-164.

Cai, X., McKinney, D., Lasdon, L., 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.

Commonwealth Scientific and Industrial Research Organization (CSIRO), 2012. Assessment of the ecological and economic benefits of environmental water in the Murray–Darling Basin. CSIRO water for a healthy country national research flagship, Australia.

Confederación Hidrográfica del Júcar (CHJ), 2014. Memoria proyecto del plan hidrológico de cuenca. MAGRAMA. Valencia.

Dagnino, M., Ward, F., 2012. Economics of agricultural water conservation: empirical analysis and policy implications. *Int. J. Water Resour. D.* 28 (4). 577-600.

Dasgupta, P., Heal, G., 1979. Economic theory and exhaustible resources. Cambridge University Press. Cambridge.

Elliot, J., Deryng, D., Müller, C., et al.,(2014. Constraints and potentials of future irrigation water availability on agricultural production under climate change. *PNAS* 111(9), 3239-3244.

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

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.

Gorelick, S., Zheng, C., 2015. Global change and the groundwater management challenge. *Water Resour. Res.* 51, 3031-3051, doi:10.1002/ 2014WR016825.

Guerrya, A., Polasky, S., Lubchenco, J., Chaplin-Kramer, R., Daily, G., Griffin, R., Ruckelshaus, M., Bateman, I., Duraiappah, A., Elmqvist, T., Feldman, M., Folke, C., Hoekstra, J., Kareiva, P., Keeler, B., Li, S., McKenzie, E., Ouyang, Z., Reyers, B., Ricketts, T., Rockström, J., Tallis, H., Vira, B., 2015. Natural capital and ecosystem services informing decisions: from promise to practice. *PNAS* 112 (24), 7348-7355.

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

Hatton, D., Morrison, M., Rose, J., Boyle, K., 2011. Valuing a multistate river: the case of the River Murray. *The Aust. J. Ag. Resour. Ec.* 55, 374-392.

Howitt, R., 1995. Positive mathematical programming. *Am. J. Agr. Econ.* 77: 329-342.

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

Intergovernmental Panel on Climate Change (IPCC), 2014. Summary for Policymakers. In: Field, C., Barros, V., Dokken, D., et al. (Eds.), Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press. Cambridge.

Martinez-Espiñeira, R., 2002. Residential water demand in Northwest of Spain. *Environ. Resour. Econ.* 21, 161-187.

McCarl, B., Dillion, C., Keplinger, K., Williams, R., 1999. Limiting pumping from the Edwards Aquifer: an economic investigation of proposal, water markets, and spring flow guarantees. *Water Resour. Res.* 34(4): 1257-1268.

McDonald, M., Harbaugh, A., 1984. A modular three-dimensional finite-difference groundwater flow model. US Geological Survey Open-File Report 83-875, Reston.

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.

Ministerio de Agricultura, Alimentación y Medio Ambiente (MAGRAMA), 2014. Anuario de aforos. Dirección General del Agua. CEDEX. Madrid.

Momblanch, A., Andreu, J., Paredes-Arquiola, J., Solera, A., Pedro-Monzonís, M., 2014. Adapting water accounting for integrated water resource management. The Jucar Water Resource System (Spain). *J. Hydrol.* 519, 3369-3385.

Richey, A., Thomas, B., Lo, M., Reager, J., Famigliettti, J., Voss, K., Swenson, S., Rodell, M., 2015. Quantifying renewable groundwater stress with GRACE. *Water Resour. Res.* 51, doi:10.1002/2015WR017349

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

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

Schewe, J., Heinkea, J., Gertena, D., et al., 2014. Multimodel assessment of water scarcity under climate change. *PNAS* 111(9), 3245-3250.

Schoups, G., Addams, L., Minjares, J., Gorelick, S., 2006. Sustainable conjunctive water management in irrigated agriculture: Model formulation and application to the Yaqui Valley, Mexico. *Water Resour. Res.* 42, W10417, doi:10.1029/2006WR004922.

Sophocleous, M., Koellikerb, J., Govindarajuc, R., Birdied, T., Ramireddygarie, S., Perkinsa, S., 1999. Integrated numerical modeling for basin-wide water management: the case of the Rattlesnake Creek basin in south-central Kansas. *J. Hydrol.* 214, 179-196.

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

United Nations Environment Programme (UNEP), 2003. Groundwater and its susceptibility to degradation. UNEP. Nairobi, Kenya.

Ward, F., Pulido-Velazquez, M., 2009. Incentive pricing and cost recovery at the basin scale. *J. Environ. Manage.* 90, 293-313.

World Water Assessment Programme (WWAP), 2006. Water: a shared responsibility. UNESCO/Berghahn, New York.

Young, R., 2005. Determining the economic value of water: concepts and methods. Resources For The Future. Washington D.C.

5.7 Appendix

5.7.1 Tables

Table B1. Intercept of the yield function (maximum yield) by irrigation district, crop and technology (Ton/ha).

Crop	Irrigation technology	Castilla La Mancha/Upstream			Valencia/Downstream			
		NEM	CEM	SEM	CJT	ESC	ARJ	RB
Rice	Flood	0.00	0.00	0.00	7.86	0.00	7.86	7.86
Wheat	Sprinkler	4.85	4.77	4.85	0.00	0.00	0.00	0.00
Barley	Sprinkler	5.25	5.22	5.25	0.00	0.00	0.00	0.00
Corn	Sprinkler	11.45	11.41	11.45	0.00	0.00	0.00	0.00
Other cereals	Flood	0.00	0.00	0.00	11.48	0.00	11.48	11.48
	Sprinkler	21.87	22.59	21.88	12.45	0.00	11.66	11.53
Garlic	Drip	8.68	8.66	8.68	0.00	0.00	0.00	0.00
Onion	Drip	92.64	92.37	92.60	0.00	0.00	0.00	0.00
Other vegetables	Flood	4.48	4.70	4.21	51.55	0.00	51.55	51.55
	Drip	5.17	5.48	4.89	54.10	0.00	52.31	51.92
Citrus	Flood	0.00	0.00	0.00	26.37	26.37	26.37	26.37
	Drip	0.00	0.00	0.00	26.95	26.54	26.56	26.46
Grapes	Drip	10.27	10.19	10.27	0.00	0.00	0.00	0.00
Other fruit trees	Flood	0.00	0.00	0.00	13.61	13.61	13.61	13.61
	Drip	2.41	2.40	2.42	14.00	13.70	13.71	13.66

Table B2. Linear term of the yield function (marginal yield) by irrigation district, crop and technology (Δ(Ton/ha)/Δha).

Crop	Irrigation technology	Castilla La Mancha/Upstream			Valencia/Downstream			
		NEM	CEM	SEM	CJT	ESC	ARJ	RB
Rice	Flood	0.00	0.00	0.00	-4.52	0.00	-0.57	-0.20
Wheat	Sprinkler	-0.74	-0.06	-0.19	0.00	0.00	0.00	0.00
Barley	Sprinkler	-0.77	-0.06	-0.17	0.00	0.00	0.00	0.00
Corn	Sprinkler	-4.64	-0.20	-0.55	0.00	0.00	0.00	0.00
Other cereals	Flood	0.00	0.00	0.00	-17.80	0.00	-12.49	-14.38
	Sprinkler	-7.15	-0.69	-1.75	493	0.00	-21.04	-133.87
Garlic	Drip	-22.13	-1.00	-3.27	0.00	0.00	0.00	0.00
Onion	Drip	-162.69	-7.82	-25.72	0.00	0.00	0.00	0.00
Other vegetables	Flood	-5.59	-0.23	-0.67	-61.37	0.00	-104.34	-272.18
	Drip	-20.33	-0.63	-1.72	-31.66	0.00	-27.68	-70.05
Citrus	Flood	0.00	0.00	0.00	-1.41	-3.19	-0.94	-2.26
	Drip	0.00	0.00	0.00	-0.40	-2.21	-0.80	-0.99
Grapes	Drip	-0.81	-0.34	-1.93	0.00	0.00	0.00	0.00
Other fruit trees	Flood	0.00	0.00	0.00	-3.74	-240.91	-21.39	-62.32
	Drip	-3.84	-0.46	-0.30	-1.77	-27.43	-2.44	-15.76

5.7.2 Mathematical appendix

The groundwater flow is calculated with a finite-difference groundwater flow equation based on the principles of water mass balance and Darcy's law. The formulation (equation 5.14) is similar to that used in the MODFLOW groundwater model (McDonald and Harbaugh 1984). Equation (14) is derived in the following way:

For simplicity and without loss of generality, we assume that n aquifer cells or sub-aquifers are represented by 1 row and n columns, where the set aqf consists of n elements: $1,2,\ldots,n$. These aquifer cells are connected serially to each other and to n river reaches, The set $river$ also consists of $1,2,\ldots,n$ elements, where every cell is connected only to one river reach. Think of the river as a multi-colored ribbon, with a separate color for each reach, flowing on top of a series of blocks below (aquifer cells) in which both the river and aquifer are divided into n contiguous cells. The water mass balance for each aquifer cell is defined by:

$$\Delta Z_{aqf,t} = R_{aqf,t} - Q_{aqf,t} + X_{aqf,t} + X_{river,aqf,t} \qquad \text{(A1)}$$

where equation (A1) states that the sum of all flows into and out of sub-aquifer, aqf, in time, t, must be equal to the rate of change in storage within that sub-aquifer, $\Delta Z_{aqf,t}$, where $R_{aqf,t}$ is the recharge of that sub-aquifer, $Q_{aqf,t}$ is the net groundwater pumping from that sub-aquifer, $X_{aqf,t}$ is the water flow between that sub-aquifer and adjacent sub-aquifers, and $X_{river,aqf,t}$ is the water flow between that sub-aquifer and the corresponding river reach.

The rate of change in storage, $\Delta Z_{aqf,t}$, in each sub-aquifer is defined as a function of the sub-aquifer head as follows:

$$\Delta Z_{aqf,t} = S_{aqf} \cdot A_{aqf} \cdot \left(H_{aqf,t} - H_{aqf,t-1}\right)/\Delta t \qquad \text{(A2)}$$

where parameters S_{aqf} and A_{aqf} are specific yield and area of that sub-aquifer, respectively. Parameter Δt is the time step, and variables $H_{aqf,t}$ and $H_{aqf,t-1}$ are the head of that sub-aquifer in the current and previous time period, respectively.

The water flow between adjacent sub-aquifers $X_{aqf,t}$ is defined by equation (A3), and the water flow between sub-aquifers and the corresponding river reaches $X_{river,aqf,t}$ is defined by equation (A4). Equations (A3) and (A4) are formulated using the Darcy's law as follows:

$$X_{aqf,t} = C_{aqf,aqf-1} \cdot \left(H_{aqf-1,t} - H_{aqf,t}\right) + C_{aqf,aqf+1} \cdot \left(H_{aqf+1,t} - H_{aqf,t}\right) \qquad \text{(A3)}$$

$$X_{river,aqf,t} = C_{river,aqf} \cdot \left(H_{river,aqf,t} - H_{aqf,t}\right) \qquad \text{(A4)}$$

where equation (A3) states that the water flows between the sub-aquifers, aqf, and adjacent sub-aquifers, $aqf-1$ and $aqf+1$, depends on the sub-aquifer heads, H, and the hydraulic conductances between sub-aquifers, C, with $X_{aqf,t}$ being negative (positive) if water is flowing out of (in) sub-aquifer, aqf. Equation (A4) states that the water flow between the sub-aquifer, aqf, and the corresponding river reach, $river$, depends on the sub-aquifer and river heads, H, and the hydraulic conductance between the sub-aquifer and the river, C, with $X_{river,aqf,t}$ being negative (positive) if sub-aquifer is discharging water to (receiving water from) the river reach.

The mass balance equation (A1) can be written using equations (A2), (A3) and (A4) as follows:

$$S_{aqf} \cdot A_{aqf} \cdot \left(H_{aqf,t} - H_{aqf,t-1}\right)/\Delta t = R_{aqf,t} - Q_{aqf,t} + C_{aqf,aqf-1} \cdot \left(H_{aqf-1,t} - H_{aqf,t}\right) + C_{aqf,aqf+1} \cdot \left(H_{aqf+1,t} - H_{aqf,t}\right) + C_{river,aqf} \cdot \left(H_{river,aqf,t} - H_{aqf,t}\right) \qquad \text{(A5)}$$

Solving for $H_{aqf,t}$ yields the groundwater flow equation (equation 5.14 in the text):

$$H_{aqf,t} = \left[1/\left\{\left(S_{aqf} \cdot A_{aqf}/\Delta t\right) + C_{aqf,aqf-1} + C_{aqf,aqf+1} + C_{river,aqf}\right\}\right] \cdot \left[R_{aqf,t} - Q_{aqf,t} + \left(S_{aqf} \cdot A_{aqf} \cdot H_{aqf,t-1}/\Delta t\right) + C_{aqf-1} \cdot H_{aqf-1,t} + C_{aqf+1} \cdot H_{aqf+1,t} + C_{river,aqf} \cdot H_{river,aqf}\right] \qquad \text{(A6)}$$

Chapter 6

Summary and conclusions

Global water resources face new challenges that suggest a renewed role for water resources modeling in the design of water management policies. Scarcity, growing populations, and massive water developments have led to keen competition over water resources. Climate change is expected to further reduce the availability of water resources and increase the variability in water supplies in some regions, especially in arid and semiarid basins. Emerging social demands for the protection of water dependent-ecosystems benefiting the society are increasing competition for already scarce water resources. Water disputes among sectors and regions are expected to increase, giving rise to complex social conflicts.

During recent decades, hydro-economic modeling is becoming an emerging tool to assist decision makers in the design and implementation of sustainable water management policies in basins. Despite the significant advancement in integrated hydro-economic modeling over the last three decades, several gaps are not yet settled in the literature, and much more progress is expected. Facing these gaps, this thesis presents the development and application of several integrated hydro-economic modeling approaches. The four main chapters of this thesis suggest selected methodological approaches to fill the gaps related to the dynamic and stochastic dimensions of hydro-economic models, and to the inclusion of the strategic behavior of stakeholders within those models.

6.1 Summary

The first article of this thesis (chapter 2) “Modeling water scarcity and droughts for policy adaptation to climate change in arid and semiarid regions”, presents the development and application of an integrated hydro-economic model for the Jucar basin. The contribution of this chapter to previous hydro-economic modeling efforts stems from the development of a reduced form hydrological component. 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 represent water flows. 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 to assess the economic and environmental effects of alternative drought management policies (policy of cooperation, water market policy, and environmental water market policy) under various drought scenarios (mild, severe, and very severe drought). The implementation of a pure water market policy in the Jucar basin show modest gains compared to the current institutional setting. In addition, 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.

The inclusion of the basin authority in the water market to acquire water for the environment seems to be an attractive policy to keep up with the basin's increasing demand for water and to correct the failure of pure water markets. This policy improves social and private benefits of the basin, reduces the vulnerability of irrigation districts to droughts, and protects ecosystems. The empirical results show the value of negotiation and stakeholders' cooperation, which is the current institutional approach to water management in Spain. This institutional approach achieves almost the same economic outcomes and better environmental outcomes compared to a pure water market policy. The policy implications of these findings highlight the importance of cooperation in water management, and call for a reconsideration of water policies used at present in most arid and semiarid regions. Current water policies are based on command and control instruments or pure economic instruments, disregarding the potential of stakeholders' cooperation and participation in decision making.

The second article (chapter 3) “Cooperative water management and ecosystem protection under scarcity and drought in arid and semiarid regions”, develops a game theory framework. This framework is used to analyze the likelihood of cooperation over sharing water resources, and the options for protecting ecosystems in arid and semiarid basins under scarcity and drought. The use of cooperative game theory to account for the strategic behavior of individual stakeholders is essential for testing the acceptability and stability of policy interventions aimed at promoting the joint management of water resources. The chapter shows how the strategic behavior of individual stakeholders could be incorporated into hydro-economic modeling.

The empirical results of this chapter indicate that drought damage costs in the Jucar basin could be reduced through the cooperation of stakeholders within the right institutional setting. The results indicate also that cooperative management may have little effect on ecosystems protection without additional incentives or regulations. The analysis performed suggests that cooperation is a feasible option in the Jucar basin, but the basis for cooperation is fragile, leading to a weak acceptability and stability of cooperative arrangements. The reason is the different interests of users among the various cooperative solutions. The internalization of environmental damages seems to provide more stability to cooperation. This finding has important policy implications because it demonstrates the difficulties in selecting a mix of policy instruments that could address scarcity and droughts in highly-stressed basins, and the risk of policy failure.

The third article (chapter 4) “Efficient water management policies for irrigation adaptation to climate change in Southern Europe”, evaluates the economic and environmental effects of two incentive-based water management policies to address climate change impacts on irrigated agriculture in Southern Europe: water markets and subsidies on irrigation technologies. These two policies are of particular interest for policymakers in Spain and in Southern Europe. The analysis is undertaken in the lower part of the Jucar basin using a two-stage stochastic hydro-economic model. Several crop-water production functions are used to allow for the modeling of deficit irrigation strategies. Model results provide information on farmers’ responses to climate change and policy interventions in terms of long-run choices of capital investment in cropping and irrigation systems, and short-run decision to irrigate or fallow land.

Results of this chapter indicate that climate change will have harmful effects on farmers’ profits. However, the severity of these effects depends on government policy settings and farmers’ adaptation responses. Water markets and irrigation technology subsidies are policies that provide good incentives to farmers for investing in irrigated agriculture, reducing land abandonment, and intensifying land and water use. In addition, the deficit irrigation strategy is found to be a valuable response to climate change, reducing significantly farmers’ damage costs. However, the environmental impacts of these policy interventions are adverse, generating considerable environmental costs for society. The policy implications of these results suggest that the European water and agricultural regulations should be oriented towards enhancing farm level adaptive capacity, improving farmers' knowledge of climate change impacts for better long-run investment decisions, and promoting the uptake of adaptation policies that minimize both private and social benefit losses.

The last article (chapter 5) "Hydro-economic modeling with aquifer-river interactions for sustainable basin management", presents the development of a dynamic hydro-economic modeling framework, which is applied to the assessment of climate change scenarios and policy choices in the Jucar basin. The contribution of this paper to previous hydro-economic modeling efforts is the incorporation of a groundwater flow equation, similar to the MODFLOW groundwater model approach, into the standard hydro-economic model formulation. This improved methodological approach is capable of simulating the real-world behavior of aquifers and the stream-aquifer interactions.

The results of this chapter demonstrate the capabilities of integrated hydro-economic models to accurately assess a wide range of climate change scenarios and policy choices. The empirical results provide detailed information on the spatio-temporal impacts of climate change on the hydrology, land use and economic outcomes. Results show also how water users in arid and semiarid basins could strategically adapt to climate change, and the economic and environmental tradeoffs among their adaptation strategies. This information can be only generated through integrated hydro-economic modeling, and it could be useful for the design of sustainable climate change adaptation policies.

6.2 Conclusions and future research

This thesis addresses some of the most important challenges facing water resources in arid and semiarid regions. These challenges are the growing water scarcity, the climate change impacts on water resources, and the pervasive degradation of water-dependent ecosystems. The main four chapters present the development of several hydro-economic modeling approaches that integrate hydrologic, economic, institutional and environmental components. These models are applied to the specific case of the Jucar basin in Spain, and the chapters provide a detailed description of the modeling process, the analysis undertaken, and the main findings and implications. However, the modeling frameworks are designed to be adaptable for other arid and semiarid basins. The methodology developed in this research constitutes a very promising tool for conducting integrated analysis of climate and policy scenarios. The obtained results could provide useful insights to policy making for sustainable water management.

This thesis considers some methods for improving the performance of water policy models. These methodological advances are related to the process of integrating the different aspects of water resources, the improvement of the stochastic and dynamic aspects of modeling, and the inclusion of the strategic behavior of individual stakeholders. Few studies in the literature address jointly the issues of both modeling and implementation of water policies. The empirical results highlight the potential of integrated hydro-economic models for assessing the environmental and economic tradeoffs among policy choices under climate scenarios. Partial modeling based only on economic relationships but without a realistic biophysical underpinning, cannot catch accurately these policy tradeoffs which are crucial for the sustainable management of water resources.

The findings in this thesis have important policy implications because they demonstrate the range of difficulties in achieving a more sustainable management of water resources in arid and semiarid regions. The sustainability challenges are how to deal with the impacts of water scarcity, droughts and climate change on the economic activities and ecosystems in river basins. Governments could implement several policy interventions to mitigate those impacts such as promoting the cooperative management of water resources, allowing for water trading, and providing economic incentives for water conservation. However, policymakers should be aware of the unintended consequences of poor policy planning, such as the third party effects of policy interventions which are found to be quite negative for the environment. They should also consider the acceptability and uptake of policy interventions by stakeholders to avoid policy failure.

For instance, the results of this thesis show that the cooperative management of water resources and the implementation of water markets could be attractive policies to reduce the economic damage costs of scarcity, droughts and climate change, but they could have devastating effects on valuable aquatic ecosystems in the absence of adequate regulation or well-functioning water institutions. In addition, the acceptability and stability of policy interventions aimed at basin-wide climate change mitigation are not evident, given the different preferences and interests of groups of stakeholders.

Overcoming these impediments is not an easy task in basins around the world. The sustainable management of water resources in arid and semiarid regions requires shifting water policies from coercive governance based on command and control instruments or from governance based on pure economic instruments, towards new governance rules based on the involvement of stakeholders coupled with the ancillary use of carefully designed economic instruments that bring about cooperation.

The results obtained in this thesis could be improved in future research work. It would be interesting to analyze the detailed impacts of regionalized climate change scenarios within a stochastic framework, especially those linked to the change in the probability distributions of key climatic variables and the occurrence of extreme events. This type of analysis could be undertaken using a hydro-economic model together with Monte Carlo simulations and the Markov switching model. Additional research might also focus on analyzing the synergies and tradeoffs between water, energy use and food production. Society is well aware of food, energy and water challenges, but researchers have so far addressed them in isolation, within sectoral boundaries. Integrated hydro-economic models could simultaneously address these challenges. A final direction for future research could be linking hydro-economic modeling to computable general equilibrium approaches. But this would have to overcome the challenge of dealing at the same time with the biophysical complexity of basins and with the whole economic activities, a quite exacting endeavor. The reward would be the capability of evaluating the economy-wide effects of policy interventions at basin scale.