ECONOMIC INSTRUMENTS FOR WATER MANAGEMENT IN SPANISH MEDITERRANEAN BASINS

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Economic Instruments for Water Management in Spanish Mediterranean Basins

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Abstract

The scarcity of adequate water has been historically a complex policy challenge in Mediterranean basins. The conventional response to this problem has focused on the construction and exploitation of water works to meet the increasing water demand and, when that was not enough, on the regulation of water demand through command and control policies. Yet, the evident technical success in harnessing the potential of water for economic growth in the past has come along with new significant challenges. Coupled with production and population growth, the demand of water services has soared up. Besides, climate-change has significantly altered water availability, giving rise to a water supply crisis which is perceived by many experts to be one of the top global risks.

Conventional policy making seemed incapable to overcome these challenges and demanded some innovations. However, path dependency, the transaction costs of policy reforms and other constraints have resulted in policy makers insisting upon conventional water policy. Supply policies have escalated, regulatory policies have become more complex (and more difficult to enforce) and the water crisis has been aggravated. Abundant evidence suggests that this policy mix has ended up increasing water demand, reducing water availability and undermining the robustness and resiliency of the system and its ability to cope with the water crisis.

Considering its failure, the longevity of conventional water policy is striking. In many areas, only the exhaustion of traditional supply sources has been able to stop it. Eventually, the financial and environmental costs of developing new water works have begun to exceed the economic benefits in the marginal uses of existing supply in many basins, and this has made conventional policies unviable. Also, budgetary constraints as a result of the financial crisis have increased the opportunity costs of water infrastructures, preventing or delaying further water works.

It is increasingly accepted that this *vacuum* in water policy needs to be filled in with innovative policies that help achieve water policy objectives through an effective and efficient management of water demand. However, achieving the collectively agreed goals of water policy through the actions of individuals is a challenging task. Experience shows that individuals with common objectives cannot be always

counted on to act voluntarily to achieve them (this explains the non-compliance problems of regulatory instruments). Therefore, the challenge is to find suitable tools that motivate collective action through the use of incentives.

Evidence in other environmental fields has shown that the best way to manage incentives is through economic instruments. Economic instruments replace the traditional notions of control and government-led planning by those of incentives, motivation and multi-level governance. If successfully combined with conventional policies, economic instruments may help to progressively overcome the current water crisis. However, developing an effective and efficient economic instrument for water management is not an easy task: whereas science has developed technical water management to a very large extent, considerations of social, political, institutional and financial order (i.e., economics) are still treated in an incipient form, with major problems persisting.

This thesis wants to help bridge this gap and presents a series of methodologies and stylized facts that are used to assess the contribution that economic instruments can make to water policy in Mediterranean basins. This is done through six scientific papers illustrated with applications to different areas located in Mediterranean river basins in Spain. First, the thesis assesses the reasons that explain the exhaustion of the traditional supply-side approach, as well as the key factors behind the rise and failure of the extensive command and control based policy that came precisely to complement it. Then, this work examines the role that economic instruments may play, in conjunction with conventional policies, in reverting the negative trends observed under the current water crisis. It is concluded that economic instruments have the potential to improve the status of overexploited water bodies, but they are not a *panacea*: an adequate design, institutional setup and policy mix are also needed to start paving the road out of the water crisis.

Resumen

La escasez de agua ha sido un reto histórico para el desarrollo de las cuencas mediterráneas. Tradicionalmente, la respuesta a este problema ha consistido en la construcción y el aprovechamiento de obras hidráulicas que permitieran incrementar la oferta y satisfacer así la creciente demanda. Cuando esto ha resultado insuficiente, la ingeniería ha venido acompañada de herramientas legales que regulaban el uso del recurso. No obstante, el evidente éxito de estas políticas convencionales en la gestión técnica del agua y en su aprovechamiento para promover el desarrollo económico de las cuencas mediterráneas ha venido acompañado de importantes desafíos. Por un lado, el crecimiento económico y el aumento de la población han incrementado la demanda de agua. Por otro lado, el cambio climático ha generado incertidumbres en torno a la disponibilidad del recurso, dando lugar a una crisis de oferta que muchos expertos sitúan entre los mayores riesgos ambientales a nivel global.

Las políticas convencionales no ofrecen suficiente capacidad de respuesta ante estos desafíos. No obstante, existen restricciones (*path dependency*, costes de transacción) que han dificultado las reformas necesarias para lograr una transición hacia una nueva política del agua. Como resultado, a menudo se ha insistido en las políticas de oferta y regulación: se han incrementado las dimensiones de las obras hidráulicas y se han aprobado normativas cada vez más complejas y ambiciosas, pero también más difíciles de hacer cumplir. La evidencia científica sugiere que esta combinación de políticas ha incrementado la demanda de agua, ha reducido la oferta y ha hecho al sistema menos resiliente y robusto, socavando su capacidad para afrontar la crisis del agua.

No obstante, este modelo de gestión ha demostrado ser sorprendentemente longevo, y solo el progresivo agotamiento de las fuentes de agua tradicionales ha frenado el desarrollo de nuevas obras hidráulicas. Como resultado de la sobreexplotación, los costes marginales de desarrollar nuevos proyectos han superado los ingresos marginales, haciendo estas políticas inviables en numerosas cuencas. Además, las restricciones presupuestarias consecuencia de la crisis financiera han incrementado el coste de oportunidad de desarrollar estos proyectos, paralizando o retrasando numerosas obras.

La gestión futura del agua pasa por un uso eficaz y eficiente de políticas de demanda innovadoras. No obstante, alcanzar los objetivos colectivos de la política del agua a través de acciones individuales no es tarea fácil. La experiencia demuestra que incluso los individuos que comparten objetivos comunes no siempre llevan a cabo de manera voluntaria las actuaciones necesarias para alcanzarlos (un claro ejemplo es el incumplimiento de numerosas regulaciones ambientales). Por tanto, el reto en la gestión del agua consiste en desarrollar instrumentos que motiven la acción colectiva a través del uso de incentivos.

La evidencia científica en otros campos de investigación ambiental ha mostrado que la mejor manera de gestionar los incentivos es a través del uso de instrumentos económicos. Los instrumentos económicos sustituyen los conceptos de regulación y toma de decisiones centralizada por los de una gestión multi-nivel, basada en la motivación, los incentivos y las decisiones voluntarias. Si se combina de manera adecuada esta herramienta con las políticas convencionales, los instrumentos económicos pueden contribuir a la solución de la actual crisis del agua. No obstante, crear instrumentos económicos eficaces y eficientes no es sencillo: mientras la gestión técnica del agua se ha desarrollado en gran medida durante las últimas décadas, la gestión social, política, institucional, ambiental y financiera del recurso (esto es, económica) se trata todavía de una manera incipiente.

Esta tesis pretende contribuir al desarrollo de instrumentos económicos que permitan una mejor gestión del agua en las cuencas mediterráneas. A través de seis artículos científicos que contienen estudios de caso en cuencas mediterráneas españolas, se presentan una serie de metodologías y hechos estilizados con dos objetivos fundamentales: en primer lugar, analizar los motivos que explican el agotamiento de las políticas convencionales, tanto de oferta como de regulación; en segundo lugar, examinar el rol que los instrumentos económicos pueden jugar complementando a las políticas convencionales en la solución de la actual crisis del agua.

Se concluye que si bien los instrumentos económicos tienen la capacidad de mejorar el estado ecológico de las masas de agua sobreexplotadas, no constituyen ni mucho menos una *panacea*: un correcto diseño, una adecuada combinación de políticas y la adaptación al marco institucional son requisitos necesarios para lograr una gestión sostenible del agua.

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Part I

General Introduction

1 Introduction

1.1 A Primer on water economics

Fresh water is a finite and vulnerable resource that is essential for sustaining life, development and the environment (ICWE, 1992). The essentialness of water is based on the fact that no production (agricultural, industrial, tertiary and even ecological) is possible if this input is lacking (Hanemann, 2006) and on the absence of any substitutive good for its final consumption (Savenije, 2002). This essentialness is relevant for economics because water is also a finite good. All water stems from rainfall, and rainfall is limited by the amount of water that circulates through the atmosphere on an annual basis. This combination explains why water cannot fully satisfy demand for its alternative uses in some areas (therefore, it is scarce) and strengthens the argument that water is an economic good (Zaag and Savenije, 2006). In fact, the scarcity of adequate water[1] to satisfy water demand is becoming the most important environmental problem in several regions worldwide, especially in those located in arid and semi-arid Mediterranean areas. In the face of a potential environmental collapse, what can we expect from economics?

Although economics has been playing an increasingly relevant role in water management since the beginning of the XXth century, water was not formally catalogued as an economic good until the Dublin Conference on Water and the Environment in 1992[2]. The fourth Dublin Principle states: “water has an economic value and should be recognized as an economic good, taking into account affordability and equity criteria” (ICWE, 1992). However, there is substantial confusion about the practical meaning of the statement that water is an “economic good” (Hanemann, 2006; Savenije, 2002; Zaag and Savenije, 2006). More shockingly, some have even asserted that there is little agreement on what this means in theory (Briscoe, 1996). In the midst of this debate, two different schools of

[1] With *adequate water* we are referring to a certain quantity of water fulfilling some quality standards.
[2] This followed up the 1977 United Nations Water Conference in Mar del Plata, Argentina, in which the water crisis was addressed for the first time at an intergovernmental level.

thought have appeared. The first school (market proponents) maintains that water should be priced through the market and that the economic value of water would arise spontaneously from the actions of willing buyers and willing sellers, therefore ensuring that water is allocated to uses that are valued highest. On the other hand, the second school defends an integrated decision making on the allocation of scant water resources, which does not necessarily involve financial transactions (Hanemann, 2006; Zaag and Savenije, 2006).

Free competition in a market is often viewed as the most efficient system for allocating resources. Therefore, if water is a commodity and the economic system in which water is used meets the preconditions for a market system, government interventions could be in principle limited to the establishment of property rights and the enforcement of contracts (Briscoe, 1996; Hodgson, 1988). From this point of view, markets can be used as a way to reallocate water from lower to higher value productive activities. For example, Chong and Sunding (2006) have shown that urban users can pay up to 10 times more for water than agricultural users. However, market economies experience shortcomings called market failures. Market failures occur when the allocation of goods and services by a free market is not Pareto-efficient, i.e., when it is possible to make a market participant better-off without making someone else worse-off. In other cases, even efficient markets may not meet societies' equity criteria and public intervention is necessary to compensate for distributional disparities. In the particular case of water, market failures tend to multiply due to the unique combination of characteristics of this resource (Hanemann, 2006; Zaag and Savenije, 2006). Apart from being essential and finite, water is also fugitive but bulky, private and public at the same time and variable along space and time. It has several roles, environmental, social and financial. Interspersed water bodies are interconnected at a basin level as part of a complex system, and therefore water uses are interrelated and affect each other. In addition and as a result of all the characteristics above, water is a heterogeneous good; this means that we cannot strictly speak about a single water market, but about different water markets.

Despite all these problems, the reasoning of market proponents has become widespread and has received the support of international institutions such as the World Bank (Briscoe, 1996). Water markets with very limited government intervention were designed in water scarce countries like Australia and Chile. Also in the EU and the US water markets have been introduced, although the regulation in these areas is more restrictive. Allegedly, the (so-called) *right price* stemming from free competition markets would encourage not only a more efficient water allocation, but also water conservation (partially through the spontaneous adoption of more advanced water technologies), lower expenditures in civil water works and the internalization of the opportunity costs of water use (Tarlock et al., 2009). However, market proponents seem to ignore a basic economic principle: market prices are not the same as economic value. In fact, observed market prices are only a rough indicator of the marginal value of water and they do not reflect non-market water uses and third-party impacts of market activities (Colby and Bush, 1987; Colby, 1987; Hanemann, 2006). This distinction between prices and economic value dates back to Adam Smith and was firstly formulated by Dupuit (1844) and Marshall (1879) and fully integrated into the economic theory in the 1970s. As a result, the advantages above are only feasible if the market is designed in such a way that prices become close to the actual value of the resource. In reality, the difficulties to make this happen are many, especially in the case of water (Hanemann, 2006). Consequently, in spite of their relative financial success, there are many examples in the literature concerning the negative environmental and economic impact of water markets (Colby, 1990; Donoso, 2011; Hanak et al., 2011; Howe, 2000; Young, 2010).

The negative environmental performance of water markets in areas such as Chile and Australia has done much damage to water economics, up to a point where some regard this field as part of the problem instead of the solution[3]. Critics argue that economics fails to internalize all the sources of value of water and therefore may be unsuitable to address the problems at stake (Savenije, 2002). However, this criticism is narrowly focused on the negative environmental performance of (some) water markets and ignores major economic achievements, such as the role of positive economics in supporting decision making, with tools such as Cost Benefit Analysis or Cost Effectiveness Analysis.

Economics is not the same as markets. While markets focus on the financial outcome, the objective of economics is to increase the total welfare or *utility*. This can be done by maximizing the total economic value and minimizing the total economic cost. Accordingly, in spite of the confusion that may arise on the practical implications of managing water as an “economic good”, economic theory is rather standard and straightforward: water economists have to look for those instruments

[3] As a result, a number of disclaimers were added to the classification of water as an economic good, stating that water is also a “social” good and that water should be affordable to the poor (Zaag and Savenije, 2006), even though this is already included in the very notion of “economic good”.

that maximize welfare from a social, environmental and financial perspective (i.e., from an economic perspective). In some cases, these instruments may be water markets and prices. In other cases, the use of water markets *per se* may be insufficient, unviable or even counter-productive, and other economic instruments (or a combination of them) can be advanced. In this complex context, economics should focus on providing the necessary information and advice for conducting an adequate decision making process. Contrary to the arguments of market proponents, economics cannot be strictly normative and should never become a substitute for policy making: it is just a tool.

1.2 Water policy and economics

For millions of years, hunters and gatherers depended on the wild plants and animals sustained by rainfall, which varied significantly from one place to another, but was on the whole insufficient to provide food for large, dense, settled populations. Over time, families began settling near springs, lakes and rivers to supply livestock and crops with water, gradually developing technologies to divert water for irrigation and domestic purposes. Many civilizations, from Babilonian to Chinese, Mayan or Roman, constructed water delivery systems such as aqueducts to carry water to cities (Hassan, 2010; Yevjevich, 1992). Although water demand continuously increased and the growing number of water infrastructures made possible the supply of increasing amounts of water, most societies were able to meet their growing water needs by capturing reliable and relatively inexpensive resources until the middle of the XXth century. Consequently, throughout all this period, comprising most of humankind's history, water management was approached primarily as an engineering problem. Water demand was in most of the places and during most of the time below the threshold that would deem water as a scarce good. Therefore, the role of economics in water policy was limited.

In the last decades, population growth and the improvement of living standards brought about by development have generated an unprecedented increase in water demand (either in agriculture, manufacturing, tourism, energy production or households), exceeding the limits of water supply for the first time in history in many areas worldwide (Molden and Sakthivadivel, 1999). In addition, climate-change induced alteration of rainfall pattern (form, intensity and timing of rainfall) has significantly modified water availability and the frequency and intensity of extreme events such as floods and droughts, up to a point where this water supply crisis is perceived by many experts to be one of the top global risks (OECD, 2013). This combination of increasing pressures over water bodies and volatile water supply are in the origin of the current water crisis. The effects of this water crisis have been particularly visible during the last years, as a result of the aggravation of the climate change and water demand trends (OECD, 2013) and especially as a consequence of poorly designed water policies since the 1950s.

Water policy since the beginning of the crisis has largely ignored the overcoming of water supply limits by water demand and has continued focusing on the construction of major infrastructures to guarantee water supply. From aqueducts, reservoirs and traditional irrigation systems, water works have escalated to inter-basin water transfers, major dams, modern irrigation devices, wastewater treatment plants and desalination plants, among others (Hassan, 2010). This is a consequence of the prevailing political consensus, which still considers that water management policies must play an instrumental role aimed at providing a package of services, which are either essential for life or strategic for the economy. Besides that, it was believed for a long time that water demand should be taken as exogenously defined outside the field of water management policy (Dinar and Saleth, 1999). In this context, the limited capacity to support the increasing water resource abstraction and discharge rates has led to a growing demand for major infrastructures and increased public support to put larger amounts of water services available to users. In turn, the positive response of water authorities to this demand has led to unrealistic expectations concerning the capacity of the system to absorb additional pressures. This perverse dynamics has ended up increasing water demand, reducing water availability and undermining the robustness and resiliency of the system and its ability to cope with the water crisis (Anderies et al., 2004; Ruttan, 2002).

As time has passed and this sort of path dependency has prevailed, water authorities worldwide have progressively found themselves facing a potential water catastrophe. Yet, managing water is a very complex societal issue that needs to involve legal, environmental, technological, financial and political considerations that are difficult to co-ordinate in an effective manner. For a long time, this complexity has often implied that the political decisions have overshadowed and prevailed over other considerations (Martin et al., 2008). In other words, the relevance of transaction costs (especially the bargaining costs required to come to an acceptable agreement with all the parties involved) has been often magnified while that of environmental costs has been reduced, thus delaying the necessary water policy reform. This follows a basic economic principle: as long as the transaction costs of the water policy reform are perceived to be larger than the opportunity costs of the *statu quo*, the former will not be implemented (Dinar and Saleth, 1999).

However, as the water crisis has been aggravated this policy framework has become difficult to sustain. Eventually, the financial and environmental costs of developing new water works have begun to exceed the economic benefits in the least productive (marginal) uses of existing supplies in many basins (Randall, 1981). In addition, tighter public budgets and especially water supply limits have increased the opportunity costs of supply policies and have made impossible to maintain the pace of investments in water works. Therefore, water authorities have been forced to alter their policy action and to focus also on water demand. This has been largely made through a more intensive use of Command and Control (C&C) policies.

C&C policies are regulatory instruments that specify a particular type of behavior that agents have to comply with. C&C policies are not new in water policy, but during the last decades they have evolved from simple rules that restricted the pressures over water bodies and that were only casually enforced to more complex and foresighted water management plans (see for example EC, 2008 and NDMC, 2013). However, the effectiveness of C&C tools is often threatened by non-compliance of water users, and this demands a high level of enforcement. Considering the powerful incentives in the economy leading to increased use of water in the short term, one of the main critiques to these policies is the expense of enforcement (or alternatively, the extensive non-compliance), especially when a complex system of rules has been developed (Pahl-Wostl et al., 2010). This is the case for example of the EU Drought Management Plans (Gómez and Pérez-Blanco, 2012).

Noteworthy, C&C policies are applied by legislation and do not use economic incentives; therefore, although C&C tools serve to control water demand, they owe little to economic theory. In reality, the role of economics in water policy until recent years has been limited and largely consisted in assessing the economic viability of projects designed by engineers. This role of economics as an assessor was also common in other areas involving projects with environmental impact. As a result, environmental economics has been largely focused during the last decades on the development and the improvement of techniques to estimate non-market (environmental) values. This initially responded to the need to put a value to non-market goods and services in Cost Benefit Analysis (CBA).

CBA is a systematic process for calculating and comparing benefits and costs of a project, decision or government policy. Since the 1950s, several valuation methods have been developed and subsequently refined to improve the results offered by CBA. Some of these methods have even moved from being "experimental" to business-as-usual (Freeman, 2003). The results obtained during all these decades have shown that environmental benefits are significant and sometimes can greatly outweigh commercial benefits (Campos et al., 2008, 2007; Hanemann, 2006), thus enhancing the role of economics in water policy.

However, environmental valuation faces many challenges[4] (Azqueta, 1994; Freeman, 2003). In particular it is feared that traditional CBA may, in its calculation of the expected net present value, attribute relatively minor importance to a possible future disaster with major economic implications. Consequently, we may have the paradoxical outcome of a project with an expected positive welfare gain turning into catastrophic losses.

Bishop (1978) remedies this by proposing that safe minimum standards are introduced unless the cost to society is unacceptably high. This precautionary principle is rapidly integrated into environmental policy and in the Rio Declaration on Environment and Development it is stated that: "In order to protect the environment, the precautionary approach shall be widely applied by States according to their capabilities. Where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation" (UN, 1992). In the case of EU water policy, the precautionary principle becomes a key element in the design of the Water Framework Directive (WFD) (EC, 2000). This directive represents a turning point in the relationship between water policy and economics in the EU.

The objective of the WFD is the attainment of a good ecological status in all surface and groundwater bodies by 2015[5]. The precautionary principle underlying the WFD

[4] The prevailing stated preferences methods have problems addressing income constraints. Also, it is not clear whether or not values respond accurately to variations in the scope or amount of the environmental good in question when a wide range of measures or policies are involved. In addition, a major challenge remains in translating sound research results into practical and understandable policy advice.

[5] In accordance to the precautionary principle (Bishop, 1978), there may be some exemptions to the general objectives that allow for less stringent objectives, extension of deadline beyond 2015, or the implementation of new projects, provided a set of conditions are fulfilled (EC, 2000).

is that water ecosystems constitute a collective heritage that must be preserved by ensuring water uses to be compatible with the preservation of these ecosystems (EC, 2000). This makes unnecessary the calculation of environmental benefits for the purpose of achieving the goals of the WFD. Accordingly, CBA is replaced by Cost-Effectiveness Analysis (CEA). CEA is a proactive decision-support tool that enables the assessment of the cost and the effectiveness of alternative policy options in realizing a preset objective (i.e., the good ecological status). In brief, it aims at identifying a combination of mitigation measures for achieving a given water policy goal at the least economic cost. It is in this setting in which economic instruments for water management develop.

1.3 Economic Instruments for water management

When the WFD was approved, it was already clear that the amount of water available in EU water bodies was not plentiful relative to water demand anymore. Water management challenges could not be solved through the capture of unclaimed water supply, like in the past, and demanded instead a sustainable water management of the existing resources. At this point, the growing marginal costs of supplying water and the increasing interdependencies among sequential water users had already generated conflicts that gave rise to the appropriation of environmental flows/stocks by private uses at high opportunity costs for the society. This tradeoff was largely tolerated by authorities until it was too evident that a collective action was needed in order to prevent irreversible effects over EU water bodies. The WFD came as a response to this need and took into account legal, environmental, political and technological considerations from the outset, while economics became the instrument to articulate all of them.

The Article 9 of the WFD introduced for the first time economic instruments for water resources management in the EU. However, it did not include any formal definition for this term. In reality, although the WFD formally opened the door for the use of economic instruments in water resources management, it focused exclusively on the role that one particular economic instrument, water pricing, might have in reducing the pressures over water bodies: “[...] water-pricing policies provide adequate incentives for users to use water resources efficiently, and thereby contribute to the environmental objectives of this directive" (EC, 2000)[6].

Consequently, with the exception of water pricing, the implementation of economic instruments for water management has not been sufficiently encouraged from EU institutions. In addition, it remained unclear what an economic instrument for water management actually is. To the best of our knowledge, the most accurate definition can be found in Strosser et al. (2013), according to which economic instruments for water management are "those incentives designed and implemented with the purpose of adapting individual decisions to collectively agreed goals (e.g. the environmental objectives of the WFD and of its "daughter" Directives)".

Following this definition, some of the economic instruments for water management implemented in the EU so far may include pollution taxes[7] (Daugbjerg and Pedersen, 2004), water use tariffs and fees (Hellegers, 2001; Miniaci et al., 2008; OECD, 1999), water load fees (ÖKO Zrt. vezette Konzorcium, 2009), water markets (Albiac et al., 2006; Calatrava and Gómez-Ramos, 2009; Garrido and Calatrava, 2009; Rey et al., 2011), voluntary agreements (Bratrich and Truffer, 2001; Gómez et al., 2013) and subsidies (Christensen et al., 2011; Institut für Umweltforschung, 2002).

According to the little evidence available, the environmental achievements of economic instruments for water management in the EU have been very limited so far. However, and this is the critical point, it is not clear whether these economic instruments actually pursued an environmental outcome and failed because of their poor design or if they were in reality a financial tool disguised as an environmental instrument to make it more acceptable (Strosser et al., 2013). In fact, those economic instruments that did not involve revenue-raising tools or that were implemented on a voluntary basis have been among the most successful of all (Bratrich and Truffer, 2001; Gómez et al., 2013). So far, the lack of *ex-post* data impedes obtaining more concluding evidence. Moreover, there is also a lack of *ex-ante* assessments and many economic instruments with the potential to encourage a more sustainable water use have not been explored up to this point. As a result, the policy discussion regarding the implementability and the expected environmental outcome of most economic instruments is based on a mix of "theoretical and less rational arguments" (Strosser et al., 2013).

[6] More recently, the EU blueprint to Safeguard Europe's Water Resources has insisted upon this idea (EC, 2012).
[7] Actually, most of these taxes are in reality tariffs (EEA&OECD, 2013).

The present work intends to shed light over this discussion. Chapters 3 and 4 develop methodologies that are used to identify relevant factors that help explain the failure of conventional supply and C&C policies in Mediterranean river basins. Chapter 5 builds on this and develops additional methods to assess the implementability and potential of some economic instruments that have been advanced in Mediterranean areas. Rather than offering a brand new water policy the goal is to explore how these economic instruments can be streamed into current water management practice in order to make a significant contribution to meaningfully solve some relevant water governance problems.

These methodologies are illustrated with applications to different areas located in Mediterranean river basins in Spain. Spanish Mediterranean river basins are a paradigmatic example of the exhaustion of conventional water policies. Combined with growing water demand and decreasing water supply, this policy failure has led to increasing scarcity and more frequent and intense droughts.

Accordingly, the methods introduced in this thesis are largely focused on water policy challenges stemming from water scarcity and droughts (irrigation modernization plans in Chapter 3, Drought Management Plans in Chapter 4 and drought insurance and water pricing in Chatper 5)[8], although they also deal indirectly with qualitative issues. In addition, there is one methodology that is used to solve water quality problems (voluntary agreements in Chapter 5).

The methodologies presented in this thesis are based on standard economics and aim to be general and applicable in basins that face water management problems similar to those experienced in the case study areas. This mostly refers to Mediterranean basins, but the methods may also be of use in several areas worldwide increasingly exposed to the water crisis (OECD, 2013).

For example, in the EU evidence shows that scarcity and droughts are not anymore a Southern European challenge (EEA, 2009). Further to scarcity, droughts have sorely increased in number and intensity throughout the EU. There have been recent drought events or threats in Portugal, Spain, southern France, Greece, Cyprus, Italy, Hungary, southeastern England and even in Germany or the North Atlantic Faroe Islands, the self-governing region of Denmark (JRC, 2013). Also in

[8] It is true that if properly designed and contextualized these policies may also help to improve the qualitative status of water bodies, but their primary goal is a different one.

Finland and the Netherlands Drought Management Plans have been approved as a response to the increasing scarcity and drought exposure (EC, 2008).

The three innovative economic instruments assessed in this thesis do fit in this policy context marked by the water crisis, and therefore could be transferred with some caveats to other geographical areas. In addition, the discussion included in this thesis is meant to feed into some of the ongoing reflections regarding water policy reform and the use of economic instruments.

2 Objectives, Thesis Outline and Research Context

2.1 Objectives

This thesis consists of six scientific papers that have been published or are under review in international journals. The thesis presents a series of methodologies and stylized facts that are used to assess the contribution that economic instruments can make to water policy in Mediterranean basins. The goal is twofold: i) first, to identify the factors explaining the failure of conventional C&C and supply policies. ii) second, to assess the viability and expected outcomes of a set of economic instruments for water management that are being implemented or whose implementation is being considered in Mediterranean basins in Spain. Associated with these two main objectives there are five specific goals:

i) Objectives related to the understanding of the failure of conventional water policies (Chapters 3 and 4)

- Chapter 3 develops an analytical framework with the objective of determining under what conditions irrigation modernization projects rebound, i.e., under what conditions farmers end up demanding and consuming a larger amount of water than before.
- The objective of Chapter 4 is to evaluate the incentives that farmers have to incur in informal groundwater abstractions after the implementation of Drought Management Plans, as well as the impact that this may have over aquifers. Some evidence for the Segura River Basin (southeastern Spain) is provided.

ii) Objectives related to the assessment of the implementability and potential of economic instruments (Chapter 5)

- Section 5.1 develops a methodology that can be used to estimate the opportunity costs of the periodical release of flushing flows in rivers whose regimes are controlled by hydropower generating facilities. This method may help to articulate voluntary agreements between water users in highly engineered rivers. Some insightful results from the implementation of this tool in the Lower Ebro (northeastern Spain) are provided.
- Section 5.2 assesses the implementability of drought insurance for irrigated agriculture through the calculation of the minimum long term cost that would be faced by a private insurance company. Some results from field experiments in the Segura and Guadalquivir river basins in Spain are provided.
- Section 5.3 develops a flexible revealed preferences model that is used to assess the impact of water pricing over agricultural water demand and consumption. Insightful results for the case study area, the Segura River Basin (southeastern Spain), are provided.

2.2 Thesis outline

This thesis is structured in four parts:

- Part I (Chapters 1 and 2) serves as an introduction and presents the objectives, thesis outline and research context of this work.
- Part II (Chapters 3 and 4) introduces the role of supply and C&C policies in water management and explores the factors that may help explain why conventional policies have failed in attaining water policy goals.
    - *Chapter 3* presents the role played by supply policies and intends to identify the reasons that help explain why they failed to solve the water crisis. In particular, it addresses the case of irrigation modernization plans such as the *Spanish Irrigation Plan* (Plan Nacional de Regadíos).
    - *Chapter 4* focuses on C&C policies, which have become increasingly relevant as a way to control water demand. This

chapter focuses on the powerful economic incentives surrounding water use and explores how these incentives may impede a sustainable water use even when C&C and supply policies are combined. The particular case of Drought Management Plans in Spain is presented.

- Part III (Chapter 5) builds on Part II and evaluates the role that economic instruments may play as a complementary policy of conventional C&C and supply policies in order to attain water policy goals.
    - *Section 5.1* evaluates the potential of voluntary agreements to contribute to the goals defined in the water policy. The case study focuses on the public-private partnership between the hydropower operator and the river basin authority to release flushing flows to improve the qualitative status of the Lower Ebro (northeastern Spain).
    - *Section 5.2* presents drought insurance for irrigated agriculture as a water saving instrument during drought events and estimates its long term cost. These costs are estimated in two Agricultural Districts in the Segura and the Guadalquivir River Basins.
    - *Section 5.3* focuses on water pricing and uses a revealed preferences model to assess the impact that higher water prices may have over agricultural water demand in the Segura River Basin (southeastern Spain).

Part II and Part III are the core of this thesis. Each chapter of the Parts II and III is backed by at least one scientific paper that addresses relevant issues for water management in Mediterranean river basins. Most of these papers have been published in international scientific journals.

- Part IV (Chapter 6) summarizes the main conclusions of the thesis along with some political recommendations that can be inferred from the work.

2.3 Research context

This thesis is the outcome of the author’s participation in different research projects at a national and EU level in the University of Alcalá and the Madrid Institute for Advanced Studies in Water Technologies (IMDEA-Water). The list of projects in which the doctoral candidate has been involved during the development of this thesis comprises a FP7 project (*Economic Policy Instruments for Water Management in Europe*), two projects developed within the *EC Freshwater Policy Framework Contract* awarded to IMDEA-Water (*Potential for Growth and Job Creation through the Protection of Water Resources, with a Special Focus on the Further Implementation of the Water Framework Directive and Floods Directive* and *Support to the various Water Framework Directive Common Implementation Strategy (CIS) groups*) and two national projects (*Contrato entre la Universidad de Alcalá y la Agrupación Española de Entidades Aseguradora de los Seguros Agrarios Combinados SA, para el estudio de las probabilidades de restricción del agua para riego en las demarcaciones hidrográficas españolas* and *Contrato entre la Universidad de Alcalá y la Agrupación Española de Entidades Aseguradora de los Seguros Agrarios Combinados SA, para la elaboración de un estudio sobre la sequía hidrológica*). Although all these projects were relevant for the development of this work, most of the research contained in this thesis was carried out within the EU's 7th Framework Contract project *Economic Policy Instruments for Water Management in Europe* (EPI-Water).

EPI-Water (Grant Agreement 265213) is a FP7 project (FP7/2007–2013) led by Dr. Jaroslav Mysiak from Fondazione Eni Enrico Mattei (Italy) and coordinated by Prof. Carlos Mario Gómez Gómez for the IMDEA-Water team, formed by himself and Gonzalo Delacámara, Miguel Solanes, Marta Rodríguez, Estefanía Ibáñez and the doctoral candidate. EPI-Water aims to assess the effectiveness and the efficiency of economic instruments in achieving water policy goals, and to identify the preconditions under which they complement or perform better than alternative policy instruments (e.g. regulatory). The work done by IMDEA-Water focused on the development of two *ex-post* case studies in the Tagus and Ebro river basins and in particular on a comprehensive *ex-ante* case study in the interconnected Tagus and Segura river basins. Within this project, the author completed two stays of two months each in the Flood Hazard Research Centre of the Middlesex University. This was partially funded by the Fórmula Santader Scholarship.

The *EC Freshwater Policy Framework Contract* (ENV.D.1/FRA/2012/0014) defines preferred suppliers with the objective of providing services to the Water Unit in DG Environment. The projects to provide these services are awarded through individual contracts in competition with the other consortiums included in the Framework Contract. The contractual period lasts for three years, with annual renovations. The Framework Contract is led by Chris Hughes from AMEC (UK) and coordinated by Prof. Carlos Mario Gómez and Prof. Gonzalo Delacámara for the IMDEA-Water team, formed by themselves and Miguel Solanes, Marta Rodríguez, Estefanía Ibáñez and the doctoral candidate.

The project *Potential for Growth and Job Creation through the Protection of Water Resources, with a Special Focus on the Further Implementation of the Water Framework Directive and Floods Directive* is a project awarded within the EC Freshwater Policy Framework Contract. The project was led by Dr. Pierre Strosser from ACTeon Environment (France) and coordinated by Prof. Gonzalo Delacámara and Prof. Carlos Mario Gómez for the IMDEA-Water team, formed by themselves and Miguel Solanes, Marta Rodríguez, Estefanía Ibáñez and the doctoral candidate. The primary objective of this study was to assess the likely impact of the protection of water resources, and in particular of the implementation of the WFD and Flood Directive, on growth and job creation.

The project *Support to the various Water Framework Directive Common Implementation Strategy (CIS) groups* is a project awarded within the EC Freshwater Policy Framework Contract. The project is led by Dr. Andrew Farmer and coordinated by Prof. Carlos Mario Gómez for the IMDEA-Water team, formed by himself and Gonzalo Delacámara, Miguel Solanes, Marta Rodríguez, Estefanía Ibáñez and the doctoral candidate. The requested services include the provision to the Water Unit in DG Environment (ENV D1) of independent, high quality and timely support and advice on scientific, socio-economic and technical issues related to the issues dealt with within the Common Implementation Strategy (CIS) of the Water Framework Directive (WFD) and Floods Directive.

The projects *Contrato entre la Universidad de Alcalá y la Agrupación Española de Entidades Aseguradora de los Seguros Agrarios Combinados SA, para el estudio de las probabilidades de restricción del agua para riego en las demarcaciones hidrográficas españolas* and *Contrato entre la Universidad de Alcalá y la Agrupación Española de Entidades Aseguradora de los Seguros Agrarios Combinados SA, para la elaboración de un estudio sobre la sequía hidrológica* were privately awarded by the Spanish Association of Agrarian Insuring Firms (Agroseguro S.A.) to the University of Alcalá. The project was coordinated by Prof. Carlos Mario Gómez Gómez. The other members of the team were Alberto del Villar, David Nortes and the doctoral candidate. The objective of these projects was the assessment of the viability of drought insurance systems in Spain.

2.4 Conferences

In the context of this thesis, the work of the doctoral candidate was presented and discussed in different conferences. These are listed below:

Insuring water: A practical risk management option in water scarce and drought prone regions? In *Frontiers in Economics of Natural Hazards and Disaster Risk Reduction - Financing Disaster Risk Reduction and Climate Adaptation*. Belpasso (Italy), 1-7 September 2013.

Simple myths and basic maths about greening irrigation. In *New Directions in the Economic Analysis of Water*. Lisbon (Portugal), 18-19 July 2013

Water efficiency and water conservation in irrigated agriculture. In *Instrumentos económicos para la gestión del agua en España*. Alcalá de Henares (Spain), 20-21 June 2013.

Simple myths and basic maths about greening irrigation. In *5th European Association of Agricultural Economists PhD Workshop*. Leuven (Belgium), 29-31 May 2013.

Myths and Maths of Water Efficiency: An Analytical Framework to Assess the Real Outcome of Water Saving Technologies in Irrigation. In *87th Annual Conference of the Agricultural Economics Society*. Warwick (UK), 8-10 April 2013.

Can Markets Save Water? Towards a Methodological Framework to Develop a Private Drought Insurance System in Semi-arid Basins: An Application to a Mediterranean Catchment. In *International Water Resource Economics Consortium (IWREC) 10th Annual Meeting*. Stockholm (Sweden), 26-31 August 2012.

Design of optimum private insurance schemes as a means to reduce water overexploitation during drought events. A case study in Campo de Cartagena (Segura River Basin, Spain). In *International Society of Ecological Economics 2012 Conference – Ecological Economics and Rio+20*. Rio de Janeiro (Brazil), 16-19 June 2012.

Do drought management plans reduce drought risk? A risk assessment model for the Segura River Basin. In *International Society of Ecological Economics*

2012 Conference – Ecological Economics and Rio+20. Rio de Janeiro (Brazil), 16-19 June 2012.

Design of optimum private insurance schemes as a means to reduce water overexploitation during drought events. A case study in Campo de Cartagena (Segura River Basin, Spain). In *Vth AERNA Conference*. Faro (Portugal), 31 May-2 June 2012.

Design of optimum private insurance schemes as a means to reduce water overexploitation during drought events. A case study in Campo de Cartagena. In *86th Annual Conference of the Agricultural Economics Society*. Warwick (UK), 16-18 April 2012.

Do drought management plans reduce drought risk? A risk assessment model for a Mediterranean river basin. In *86th Annual Conference of the Agricultural Economics Society*. Warwick (UK), 16-18 April 2012.

Design of optimum private insurance schemes as a means to reduce water overexploitation during drought events. A case study in La Campiña (Guadalquivir River Basin, Spain). In *The Governance of Sustainability*. Cambridge (UK), 11-12 April 2012.

Development of private insurance schemes as a means to reduce water overexploitation during drought events. A case study in Campo de Cartagena (Segura River Basin, Spain). In *123rd European Association of Agricultural Economists Seminar. Price volatility and farm income stabilization*. Dublin (Ireland), 23-24 February 2012.

2.5 Publications

Chapter 3 gave rise to the following paper:

Gómez, C.M., Pérez-Blanco, C.D. (under review). Simple Myths and Basic Maths about Greening Irrigation. Under review.

Chapter 4 gave rise to the following paper:

Gómez, C.M., Pérez-Blanco, C.D., 2012. Do drought management plans reduce drought risk? A risk assessment model for a Mediterranean river basin. Ecological Economics 76, 42–48.

Chapter 5 gave rise to the following papers:

Pérez-Blanco, C.D., Gómez, C.M., 2013. Designing optimum insurance schemes to reduce water overexploitation during drought events: a case study of La Campiña, Guadalquivir River Basin, Spain. Journal of Environmental Economics and Policy 2, 1–15.

Gómez, C.M., Pérez-Blanco, C.D., Batalla, R.J., 2013. Tradeoffs in river restoration: Flushing flows vs. hydropower generation in the Lower Ebro River, Spain. Journal of Hydrology (in press, available online at http://www.sciencedirect.com/science/article/pii/S0022169413006161)

Pérez-Blanco, C.D., Gómez, C.M. (forthcoming). Insuring water: A practical risk management option in water scarce and drought prone regions? Accepted for publication in *Water Policy*.

Pérez-Blanco, C.D., Delacámara, G., Gómez, C.M. (under review). Water pricing and water saving in agriculture. Insights from a Revealed Preferences Model in a Mediterranean basin. Under second review in *Agricultural Water Management*.

Other peer reviewed publications of the doctoral candidate include:

Gutiérrez, C., Pérez-Blanco, C.D., Gómez, C.M., Berbel, J. (forthcoming). Price Volatility and Water Demand in Agriculture. A Case Study of the Guadalquivir River Basin (Spain). Accepted for publication in Bournaris, T., Berbel, J., Manos, B., Viaggi, D. (Eds.), *Economics of Water Management in Agriculture*. Science Publishers.

Pérez-Blanco, C.D., Gómez, C.M., del Villar, A. (2011). El riesgo de disponibilidad de agua en la agricultura: una aplicación a las cuencas del Guadalquivir y del Segura. *Estudios de Economía Aplicada* 29 (1):333-358

Pérez-Blanco, C.D., Gómez, C.M., Garrido-Ysete, R. (2010). Cambio estructural regional y agua: escasez, dependencia e impactos sobre el tejido económico. El caso de Andalucía. *Estudios de Economía Aplicada* 28 (2):423-446

Part 2

Supply and Command and Control Policies for Water Management

3 Supply policies for water management

In spite of the substantial advances made by water economics in the last decades, the comprehension of water management still seems largely influenced by the archaic views of John Locke and Francis Bacon, according to which “nature is only subdued by submission”. Water policies worldwide, just like those of a couple of millennia ago, are usually approached as an engineering problem and are still intensive in physical capital. Consequently, since the beginning of the water crisis the scarcity of adequate water has been battled through the construction of major infrastructures. However, there is abundant evidence that shows that supply policies, even if combined with C&C policies, are unlikely to achieve a sustainable water use unless they are complemented with (carefully developed) economic instruments (Bratrich and Truffer, 2001; Gómez et al., 2013; Strosser et al., 2013). Moreover, in many cases conventional water policies have generated unrealistic expectations on the capacity of the system to absorb additional pressures and have ended up increasing water demand and aggravating the water crisis.

Among all the Mediterranean countries, Spain is a paradigmatic example of water management captured by supply policies. For many decades, economic growth in its water scarce and drought prone Mediterranean basins has been closely linked to the capacity of public institutions to make increasing amounts of water available to users. As a result, the main strategy followed by river basin authorities consisted in coordinating the public effort required to supply the water services demanded as a result of advances in the many areas of the economy, including population growth, urban sprawl, irrigation development, growing manufacturing activities, etc. The main objective of water policy, therefore, consisted in finding inexpensive and reliable means to meet water demand. Nonetheless, as early as the 1980s it was acknowledged that water demand had started to overcome water supply and some basins were declared overexploited (BOE, 1986). As a result, this supply-oriented *modus operandi* is currently under transition to a new one aimed at making all water services used by the economy consistent with the preservation and adequate protection of the water bodies (i.e., to decouple growth from increases in water supply). However, even during this transition period some large water works have been built and others have been expanded. This includes dams, water transfers, subsidies to drill wells, the modernization of transportation, distribution and irrigation networks and more recently the development of non-conventional water sources, including treated wastewater and, especially, desalinated water.

Spain is among the countries with the most regulated rivers in the world. Total dam storage capacity has been multiplied by 50 since the 1920s and now equals 55 324 million cubic meters (or cubic hectometers, hm^3), 51% of the average annual runoff of 109 488 hm^3 (EEA, 2009; MAGRAMA, 2013a). This continued large investment has gradually reduced the marginal return of each project. For example, Gómez (2009) showed that while the total hydropower installed capacity grew by 145% from 1928 to 2012, average hydropower production remained very similar. It seems therefore that these water works aim at granting a minimum output and reducing variability instead of making additional profits.

Water policy in Spain is also rich in water transfers. Both intra and inter-basin water transfers have been implemented for centuries, although the latter usually generate the most relevant socioeconomic conflicts (ERBA, 2008; SRBA, 2008; TRBA, 2008). Only the Ebro River Basin comprises nine inter-basin water transfers, of which two of them supply water to the urban areas of Tarragona and Bilbao (ERBA, 2008). However, the largest and also the most conflictive water transfer in Spain is the Tagus-Segura Water Transfer, a major diversion project with the capacity to transfer up to 1 000 hm^3/year to the Segura River Basin from the Tagus headwaters located 242 km away. Since its opening in 1978, this infrastructure has nonetheless been working much below its legal capacity of 600 hm^3/year and has transferred in average 329.3 hm^3/year (SRBA, 2008); in addition, it is said to have passed on water scarcity problems to the Tagus River Basin (TRBA, 2008) and has been the cause of a major conflict between the regions of Castile-La Mancha (NUTS2: ES42) (largely belonging to the Tagus River Basin District) and Murcia (ES62) (largely belonging to the Segura River Basin District). The failure of this transfer to deliver its expected outcome is one of the reasons explaining the derogation of the polemic Ebro Water Transfer, which projected the transfer of 1 050 hm^3/year of water from the Ebro River Basin to various areas in the Spanish Mediterranean coast in the framework of the *Spanish Hydrological Plan* (MIMAM, 2000).

The Ebro Water Transfer was substituted by the A.G.U.A. Programme (standing for *Actuaciones para la Gestión y la Utilización del Agua*, or *Actions for the*

Management and Use of Water). This initiative also projected an increase of 1 050 hm^3/year in water supply in water scarce Mediterranean areas, but this time mostly through the use of desalination plants. The programme forecasted the construction of 20 desalination plants and the modernization of the existing ones at a cost of €3 900 million (BOE, 2005a). Since A.G.U.A. relied on saltwater instead of freshwater, this project did not put additional pressures over continental water bodies. However, while the (financial) cost of conventional water sources in Mediterranean areas is in average below 0.1 €/m^3, that of desalinated water is around 1 €/m^3 (Maestu and Villar, 2007). As a result, desalinated water needed to be heavily subsidized in order to guarantee the minimum demand that would make desalination plants financially viable (desalinated water for agricultural production in southeastern Spain is sold at 0.36 €/m^3)[9]. Even so, desalination capacity is being used below 20% in many areas (SRBA, 2008). More recently, the high financial costs of these plants and the budgetary constraints resulting from the financial crisis have increased the opportunity costs and threatened the viability of this programme. In 2012 the Spanish Ministry of Agriculture and Environment negotiated a €500 million loan used to rescue the public water company that manages the production and supply of desalinated water in Southeastern Spain, *Acuamed*. In 2013, further budgetary pressures have forced public institutions to negotiate an additional €700 million loan (GWI, 2013).

Other relevant supply policies comprise the subsidies for drilling new wells (which triggered the aquifer depletion witnessed in many areas in southeastern Spain since the 1960s) (Sevilla et al., 2010), and wastewater treatment plants (which nonetheless are able to provide a very limited supply, estimated between 50-60% of the urban water demand) (Maestu and Villar, 2007). But the most ambitious supply policy implemented in Spain in recent years has been the irrigation modernization project known as *Plan Nacional de Regadíos* 2000-2008 (*National Irrigation Plan,* PNR).

Irrigated agriculture has historically played a key economic role in Mediterranean rural areas. Even today, irrigation represents the only real pathway towards development in many of these areas. Without irrigated agriculture, these areas would face depopulation, the abandonment of the land (with potentially negative environmental effects) and a great imbalance in the population distribution. Water

[9] This is not to say that the use of desalinated water is not viable. Actually, the total economic costs of conventional water sources in overexploited areas of the Southeast are likely to be very close to (or even above) those of desalinated water (SRBA, 2008).

scarcity, being especially intense in the most productive agricultural districts of the Mediterranean, is currently the main threat to the subsistence of irrigated agriculture[10] (OECD, 2010). The Spanish PNR aimed to help the irrigation-based agriculture to face all these challenges.

The PNR invested €7 368 million over the period 2000-2008 to modernize 2 244 570 ha of irrigated lands across Spain, and planned a reduction in water withdrawals of 3 662 hm^3/year (MAGRAMA, 2013b). From a technical perspective, this project achieved an overwhelming success: while in 2002 gravity irrigation represented 40.5% of irrigated surface, drip irrigation 34.3% and sprinkler irrigation 18.4% (other irrigation systems accounted for 6.9% of irrigated lands), by 2009 the more efficient drip irrigation already represented 48.6% of irrigated surface, replacing both gravity (32.3%), sprinkler (15%) and other irrigation systems (4.1%) (Lopez-Gunn et al., 2012; MAGRAMA, 2013b).

However, according to some *ex-post* evidence, the PNR did not perform as expected in terms of water savings/conservation (Corominas, 2010; Gutierrez-Martin and Gomez, 2011; Rodríguez-Díaz et al., 2012). There are some reasons that may explain this outcome. First of all, the most relevant opportunities to save water through technical efficiency gains[11] in Spain are located in areas where water is less scarce and therefore less valuable. In turn, the opportunities to save valuable water are increasingly located in places where shifting to more efficient devices is not profitable due to the higher operation costs (e.g., energy demands) and more complex management practices (Corominas, 2010). On top of that, it has been shown that after an irrigation modernization water demand may "rebound" (i.e., increase). According to some authors, this may be the case of some areas in

[10] This menace has been recently aggravated with other problems, including decreasing subsidies, compliance with a more restrictive environmental legislation, price instability and, at the same time, the need for farmers to produce competitive goods in a global market.

[11] Water is often linked to the idea of efficiency. Nonetheless, water efficiency is a rather vague concept that needs further clarification. Specifically, it is important to differentiate between technical and economic or allocative efficiency. Up to this point, with "efficiency" we were referring to the allocative efficiency concept. In economics, the allocative efficiency is reached when the social surplus is maximized with no deadweight loss, i.e., when the value that the society assigns to the outputs produced is larger than the value that the society assigns to the inputs consumed. Here, though, we are referring to the technical efficiency, i.e., the effectiveness with which inputs are used to produce an output, or alternatively the ratio of outputs (in economic terms) to inputs (water use). Although the difference may seem subtle, it does matter. For example, better irrigation technologies increase technical efficiency, but they do not guarantee steady or declining resource use. As a result, rebound effects may appear, offsetting the technical efficiency gains and possibly generating an allocative ineffiency.

southern Spain (Gutierrez-Martin and Gomez, 2011; Rodríguez-Díaz et al., 2012). This inevitably raises the following question: why would water demand increase after an increase in the technical efficiency?

Water is fugitive and ultimately flows under gravity (Zaag and Savenije, 2006). Since water is rarely fully consumed by a single user[12], what is left can be reused by another user located downstream. This implies that a single molecule of water can have multiple and sequential uses, thus generating multiple and sequential economic values. However, it is very difficult to keep track of every single molecule of water, and consequently it is difficult to enforce property rights over return flows. If technical efficiency is increased, consumptive uses also increase and the whole water dynamics may be significantly altered. Authors have identified two effects that may result in a "rebound" of irrigation modernization plans like the PNR: an increase in water use due to the shift to more water intensive crops and the increase in use during dry periods, known as the *Jevons' Paradox* (Gómez, 2009); and an increase in water depletion due to lower water returns, known as the *Hydrological Paradox* (Pfeiffer and Lin, 2012; Rodríguez-Díaz et al., 2012; Ward and Pulido-Velazquez, 2008). While the latter is widely known and cited in the scientific literature, there is no methodological framework that identifies under what conditions a *Jevons' Paradox* will appear.

The next paper, prepared by the doctoral candidate and the Prof. Carlos Mario Gómez Gómez, presents an analytical framework that identifies the preconditions under which a *Jevons' Paradox* may occur. The paper is entitled *Simple Myths and Basic Maths about Greening Irrigation*, and is currently under review. Previous versions of this paper have been presented in the *87th Annual Conference of the Agricultural Economics Society* in Warwick, UK (8-10 April 2013); in the *5th European Association of Agricultural Economists PhD Workshop* in Leuven, Belgium (29-31 May 2013); and in the conference *New Directions in the Economic Analysis of Water* in Lisbon, Portugal (18-19 June 2013).

[12] Water use refers to the amount of water demanded by users. The share of water use that either evaporates or becomes contaminated is consumed (i.e., water consumption) (Kohli et al., 2010). The water that remains in the system can still be incorporated into other water use/s.

Simple Myths and Basic Maths about Greening Irrigation

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Abstract: Greening the economy is mostly about improving water governance and not only about putting the existing resource saving technical alternatives into practice. Focusing on the second and forgetting the first risks finishing with a highly efficient use of water services at the level of each individual user but with an unsustainable amount of water use for the entire economy. This might be happening already in many places with the modernization of irrigated agriculture, the world's largest water user and the one offering the most promising water saving opportunities. In spite of high expectations, modern irrigation techniques seem not to be contributing to reduce water scarcity and increase drought resiliency. In fact, according to the little evidence available, in some areas they are resulting in higher water use. Building on basic economic principles this study aims to show the conditions under which this apparently paradoxical outcome, known as the *Jevons' Paradox*, might appear. This basic model is expected to serve as guidance for assessing the actual outcomes of increasing irrigation efficiency and to discuss the changes in water governance that would be required for this to make a real contribution to sustainable water management.

Keywords: Jevons' Paradox; Rebound effect; Agricultural economics; Water economics; Irrigation efficiency.

JEL classification: Q15, Q18, Q25, Q51, Q58.

1. INTRODUCTION

Climate change, water supply limits, continued population growth and the improvement of living standards brought about by development are making water scarcity one of the most pressing environmental problems worldwide. Among all the competing uses, agriculture is the world's largest water user and is often believed to be wasteful. Consequently, policy makers have recently called for measures to save water in this sector. Among these measures, subsidies to increase irrigation efficiency (or technical efficiency, i.e., the effectiveness with which water is used at a plot level to produce agricultural goods) have rapidly become widespread (OECD, 2008)[1]. It is widely believed that more efficient irrigation technologies save water, making it available for other productive uses and also for the environment. However, technical options to reduce water use and withdrawals[2] are but a social opportunity that might be wasted if no other measures necessary to improve water governance are set (e.g., enforcing property rights, water pricing and metering, etc.). In fact, recent empirical work shows that even when the desired technical shift is successfully implemented, it might end up reinforcing the already unsustainable trends in water use (Pfeiffer and Lin, 2012; Rodríguez-Díaz et al., 2012). There are two arguments that help to explain this apparently paradoxical outcome: i) the *hydrological paradox*, based on the hydrological assessment of irrigation efficiency increases; and ii) the *Jevons' Paradox*, grounded on economic theory and on which the present work focuses.

The *hydrological paradox argument* comes from the hydrological study of the water balance[3] within a basin. Take for example a traditional irrigation system. Due to its low technical efficiency, a large share of the water used does not effectively contribute to satisfy evapotranspiration (i.e., the consumptive use of water or water consumption) and is therefore "lost". But much of this water is later on recaptured and returned to the watercourse, and is still available for alternative uses. However, after an increase in the irrigation efficiency, although water use may actually fall, water availability for other uses may decrease through increased consumptive use, reduced return flows and lost aquifer seepage. This *hydrological paradox* can be found for example in Jensen (2007), Rodríguez-Díaz et al. (2012), Scheierling et al. (2006), Ward and Pulido-Velazquez (2008).

The *Jevons' Paradox argument* comes from the economic study of water: without any complementary policy, an increase in irrigation efficiency makes water a more productive input

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[1] Government subsidies for irrigation modernization are common across OECD countries, covering the totality or part of the irrigation modernization costs. This is the case for example of Australia, Austria, Mexico, the Netherlands, Portugal or Spain (OECD, 2008).

[2] Water withdrawal is water removed from its source for a specific use, while water use refers to the amount of water demanded by users. The two flows are not the same because of leaks. In this paper we assume that there is no change in the transportation efficiency and we will refer directly to water use.

[3] In hydrology, a water balance equation can be used to describe the flow of water in and out of a system. A general water balance equation is:

$$P = Q + E + \Delta S$$

Where P is precipitation, Q is runoff, E is evapotranspiration and ΔS is the change in water storage (in soil or the bedrock).

and may result in an increase, rather than a reduction, in water use[4]. The idea that under certain conditions an increase in the irrigation efficiency may lead to an increase in the use of a resource is well known in economics at least since the XIXth century and has received different names, such as the Khazzoom-Brookes Effect or the *Jevons' Paradox* (Alcott, 2008, 2005; Khazzoom, 1989). There is considerable interest in determining under what conditions this paradox appears, and much research is ongoing in fields such as energy or transportation (Brookes, 1990; Greene et al., 1999; Hong et al., 2006; Vringer et al., 2007). Surprisingly, its study in the field of water economics is relatively new and mostly based on *ex-post* empirical results (Ding and Peterson, 2006; Gutierrez-Martin and Gomez, 2011; Lecina et al., 2010; Pfeiffer and Lin, 2012). There is no methodological framework that explains under what conditions an increase in the irrigation efficiency will result in a *Jevons' Paradox*. Therefore, it is difficult to predict *ex-ante* the impact that an increase in the irrigation efficiency will have over water use. This knowledge gap is shocking if we consider the prominent role that has been assigned to the modernization of irrigation devices in drought and water scarcity strategies worldwide, as well as the high costs of these projects in a time of financial crisis. This paper wants to help bridge this gap. In the following pages we present an analytical framework to discriminate the determinants of the emergence of the *Jevons' paradox*. This study may serve as a methodological guidance for empirical papers analyzing the issue.

2. ANALYTICAL FRAMEWORK

The change in water use following an increase in the irrigation efficiency depends on three opposing effects, namely, a *technical effect*, a *cost effect* and a *productivity effect*.

First of all, an increase in irrigation efficiency will reduce the amount of water required to obtain the same products as before (*technical effect*). Accordingly, water use could be reduced in the same percentage as the increase in irrigation efficiency, provided that the farmers obtain the same crops as before. This over-simplistic scenario, where no other effects are considered, is the hidden assumption of many studies assessing the expected water savings from irrigation modernization plans (a good example of this can be found in the Spanish Irrigation Plan[5]). However, the technical shift means also a change in the incentives in place and farmers will not normally continue producing the same products as before. Two additional effects over water demand need to be considered.

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[4] There is a third possibility: neither an increase, nor a decrease, but rather *no change*. That is, the same amount of water is used as before; none is saved for (1) other uses or (2) the natural environment. For the purpose of rejecting irrigation efficiency increases as a water-saving measure it is enough to show that there is no change in water use, i.e. no savings.

[5] The Spanish Irrigation Plan (*Plan Nacional de Regadíos*, PNR) 2000-2008 was a large investment effort with the aim of reducing agricultural water use. This Plan was complemented with the Shock Plan 2006. Both plans invested 7,368 M EUR to modernize 2,244,570 ha of irrigated lands and forecasted a reduction in water use of 3 662 hm3/year (MAGRAMA, 2013). However, since the implementation of the PNR, water use from agriculture in these areas are far from decreasing (Gutierrez-Martin and Gomez, 2011; Rodríguez-Díaz et al., 2012).

The second effect stems from the higher water application costs associated to more efficient irrigation technologies and, similar to the first one, reduces water use (*cost effect*). The increase in the water application costs is largely explained by the higher energy costs of the more sophisticated irrigation devices (e.g., drip irrigation) as compared to traditional devices (e.g., gravity irrigation) (Soto-García et al., 2013). For example, recent empirical work has found that the intense irrigation modernization in Spain has increased energy consumption in irrigated agriculture by 1,800% since 1950 (Corominas, 2010).

The third effect refers to the fact that more efficient irrigation systems make water more productive (*productivity effect*). Therefore, for a given amount of water use, the last drop generates a larger agricultural product than before and, for this reason, farmers would probably be willing to use more water than before. This productivity effect may have a large impact over water use and result in a *Jevons' Paradox*, though it has been traditionally ignored in the assessment of irrigation modernization plans.

Summing up, the increase in irrigation efficiency leads to three different effects making possible to obtain the same production with less water use and higher water application costs, but also with higher water productivity. The relevant question we want to solve is what would be the combined effect of the technical shift over water use. In other words, under what conditions an increase in the technical efficiency with which water is used in agriculture will lead to an increase in water use and therefore to a *Jevons' Paradox*. In order to answer this question, we develop a methodology in two stages: in the first one we obtain water demand as a function of irrigation efficiency; in the second one, we assess the impact of an increase in the irrigation efficiency over water use, identifying the determinants of the incidence of each of the three effects above.

2.1. The water demand function

Water used by farmers (W) is bought at a unitary price, *P*, (for example, per cubic meter of water used) and applied to the crops incurring in a unitary water application cost, $c(E)$. Therefore, the marginal cost of water use ($MC\ (W)$) is equal to:

$$MC\ (W) = P + c(E) \qquad [1]$$

Where $c(E)$ is an increasing function of the technical efficiency E ($E \in [0,1]$) of the irrigation devices in place[6], since more sophisticated techniques are costlier ($c'(E) > 0$) (Corominas, 2010; Soto-García et al., 2013).

The amount of water that effectively satisfies the agronomic water needs of the crops, or water consumption (EW), is only a fraction E of the total water use (W). Therefore, to consume one unit of water, farmers use $\frac{1}{E}$ units of water. Accordingly, the marginal cost of the water consumed ($MC\ (EW)$) is equal to:

[6] E measures the technical efficiency of the irrigation technology in place with, for example, typical values of 0.5 for traditional gravity, 0.7 for sprinklers and 0.9 for drip devices.

$$MC\ (EW) = \frac{P+c(E)}{E} \qquad [2]$$

Water consumption serves to produce crops (Y) with a decreasing marginal productivity:

$$Y = f(EW), \text{ with } f'(EW) > 0 \text{ and } f''(EW) < 0 \qquad [3]$$

Farmers will demand water up to the point where the marginal productivity of the water consumed ($f'(EW)$) equals its marginal cost ($MC\ (EW)$):

$$f'(EW) = \frac{P+c(E)}{E} \qquad [4]$$

Accordingly, the water demand function can be expressed as:

$$P = f'(EW)E - c(E) \qquad [5]$$

2.2. What happens with water use after an increase in the irrigation efficiency?

The answer to this question lies formally on the response of water use (W) to an increase in the irrigation efficiency (E), that is to say, on the sign of the following derivative:

$$\frac{\partial W}{\partial E} \qquad [6]$$

A positive sign (i.e.,$\frac{\partial W}{\partial E} > 0$) means that more water is used after an increase in the irrigation efficiency, and thus that the *Jevons' Paradox* applies.

Provided that following the irrigation efficiency increase there is no complementary pricing policy and thus water prices remain constant ($\Delta P = 0$)[7], the effect of an increase in the irrigation efficiency over water use can be obtained from the demand function [5] as follows:

$$\frac{\partial P}{\partial W} dW + \frac{\partial P}{\partial E} dE = 0 => \frac{dW}{dE} = -\frac{\partial P/_{\partial E}}{\partial P/_{\partial W}} \qquad [7]$$

That is to say:

$$\frac{dW}{dE} = -\left(\frac{W}{E} + \frac{f\prime(EW)}{f''(EW)E^2} - \frac{c'(E)}{f''(EW)E^2}\right) \qquad [8]$$

Which, after multiplying both sides by $\frac{E}{W}$, can be transformed into the efficiency elasticity of water use ($\epsilon_{W,E}$):

$$\epsilon_{W,E} = -1 - \frac{1}{\epsilon_{f\prime,E}} + \left(\frac{\epsilon_{c,E}}{\epsilon_{f\prime,E}}\right)\left(\frac{c(E)}{P+c(E)}\right) \qquad [9]$$

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[7] This is the case in most of the irrigation modernization plans, such as those of Spain, Portugal, Mexico or Australia (OECD, 2008).

Where:

$\epsilon_{WE} = \frac{\partial W}{\partial E}\frac{E}{W}$ is the efficiency elasticity of water use.

$\epsilon_{c,E} = \frac{\partial c(E)}{\partial E}\frac{E}{c(E)} = c'(E)\frac{E}{c(E)} > 0$ is the efficiency elasticity of the water application cost.

$\epsilon_{f\prime,E} = \frac{\partial f\prime(EW)}{\partial E}\frac{E}{f\prime} = \frac{f''(EW)}{f\prime(EW)}EW < 0$ is the efficiency elasticity of the marginal productivity of water consumption.

Equation [9] contains the three effects identified above, namely:

- A *technical effect*, meaning that increasing irrigation efficiency by one percentage point would reduce water use by one percentage point, a reduction in water use proportional to the relative increase in irrigation efficiency (indicated by -1).
- A *cost effect*, meaning that the higher application cost of water resulting from a more efficient irrigation technique will lead to a reduction in water use. This is measured by $\left(\frac{\epsilon_{c,E}}{\epsilon_{f\prime,E}}\right)\left(\frac{c(E)}{P+c(E)}\right) < 0$. The incidence of this effect over water use depends on two ratios: the first ratio $\left(\frac{\epsilon_{c,E}}{\epsilon_{f\prime,E}}\right)$ is the quotient of the efficiency elasticity of the water application cost ($\epsilon_{c,E}$) to the efficiency elasticity of the marginal productivity of water consumption ($\epsilon_{f\prime,E}$); and the second ratio $\left(\frac{c(E)}{P+c(E)}\right)$ is the quotient of the application cost $c(E)$ to the unitary water costs ($P + c(E)$).
- A *productivity effect,* meaning that the increase in water productivity will lead to an increase in water use. This is measured by $-\frac{1}{\epsilon_{f\prime,E}} > 0$ and its importance depends on the vale of $\epsilon_{f\prime,E}$.

3. CONCLUSIONS

In this paper we have developed an analytical framework that may be used to predict *ex-ante* the likelihood of a Jevons' Paradox in irrigated agriculture. This study may serve both as guidance for future empirical papers and as an analytical framework to better understand the opposing effects existing behind an increase in the irrigation efficiency.

Subsidies to increase irrigation efficiency have rapidly become a widely used policy in water stressed countries as a means to reduce water use. However, the common belief that considers more efficient irrigation devices as synonymous of water saving technologies is rather naive as it tends to ignore the entire physical, economic and institutional framework where these alternatives are implemented. Actually, making a real contribution to reduce water pressures out of a technical shift is a difficult task. To start with, water availability may decrease through increased consumptive water use, reduced return flows and lost aquifer seepage, leading to a

Hydrological Paradox. More importantly, if the *productivity effect* resulting from a better irrigation technology is large enough, total water use may even increase (*Jevons' Paradox)*.

Consider for example the following extreme, but still likely, case. Assume an agricultural area where energy is heavily subsidized and the more efficient irrigation devices do not increase the cost of applying water (for the sake of the argument, let us assume that $\epsilon_{c,E} = 0$). In addition to that, water is scarce in such a way that most of the time there is idle irrigation capacity and the technical shift will allow higher water consumption at the same cost as before ($\left|\epsilon_{f\prime,E}\right| < 1$). Then the productivity effect is higher than one $\left(-\frac{1}{\epsilon_{f',E}} > 1\right)$ and will overcome the technical effect (-1). In such a situation increasing the irrigation technology will lead to a *Jevons' Paradox* and, contrary to the common belief, water availability will decrease and the real outcome of the presumed water saving technologies will worsen the already unsustainable use of water. The intuition behind the example shows that water technologies might be less effective precisely in the situations where water savings are more needed; that is to say, in water stressed areas with subsidized infrastructures and low water and energy prices (this is the case in many Mediterranean countries like Australia, Portugal or Spain).

If the policy goal is simply to use or consume less water for irrigation, there are other measures for doing this. They are direct, inexpensive and by definition effective: caps and/or taxes. However, the actual objective of irrigation modernization policies is twofold: reducing water use without impairing agricultural welfare. More efficient technologies may help to attain this dual objective, but they should not be used as a panacea and need to be part of a comprehensive policy mix towards a sustainable water management. For example, the technical shift can increase farmers' income, and this may be used as an opportunity to agree upon a reduction in energy subsidies and/or the implementation of metering and volumetric tariffs. This policy mix, rather than a simple technical shift, can find the way to make the reduction of water scarcity compatible with the maintenance and eventual improvement of farmers' welfare. Technical options are only opportunities; the real challenge in the transition towards a sustainable water use relies on building better institutions and putting effective incentives in place.

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4 Command and Control policies for water management

In the last few decades, water resources management has become a problem that can no longer be treated exclusively from the technical standpoint (i.e., through water supply policies). The unparalleled increase of water withdrawals has forced policy makers to implement some measures concerning also water demand. This has been done through a more intensive use of Command and Control (C&C) policies. C&C policies are not new in water policy, and they have been used as a complement to supply policies for centuries. However, in the last decades they have evolved from simple rules that were only casually enforced by law to more complex and sophisticated water management plans, fully supported by regional, national and/or supra-national institutions (Pahl-Wostl et al., 2010).

The mechanism called C&C refers to a set of regulatory instruments that specify a particular type of behavior that agents have to comply with. Traditionally, the intended behavior was decided unilaterally and enforced by legal disciplining through the use of the power of the state. More recently, many institutions worldwide have opted for a more integrative approach in which the objectives of the C&C policies are designed through a social agreement including all the agents affected, although decisions are ultimately enforced by the public sector (EC, 2008, 2007; NDMC, 2013).

In the EU, complex and sophisticated C&C policies have proliferated during the last decades. Initially, C&C policies at a Community level aimed towards defining minimum standards for water quality across the EU. We can identify two milestones: the Drinking Water Directive (EC, 1991) and the Wastewater Directive (EC, 1998). Although they have performed relatively well, their implementation has been (and still is) challenging mainly because of the financial and planning aspects related to major infrastructure investment such as sewerage systems and water treatment facilities (EC, 2013; KWR, 2011). More recently, C&C policies have also addressed the challenges posed by floods (EC, 2007) and droughts (EC, 2008). Although it is still early to assess the outcomes of these policies, some authors have already claimed that these policies should be defined in a less rigid, more adaptive way to be successful in the long and even in the medium term (Pahl-Wostl et al., 2010). At a national level, C&C policies have also played a significant role, either through the transposition of Community Directives or through the development of complementary C&C policies. For example, in the overexploited Segura River Basin in Spain the allocation of new water rights for irrigation was prohibited by Royal Decree in 1986[13] (BOE, 1986). Another example can be found in the rigid norms that regulate the Tagus-Segura Water Transfer (BOE, 2001, 1985, 1980).

From the perspective of an omniscient and omnipresent state, C&C policies appear enough to achieve the objectives of water policy. However, in reality the effective implementation of these regulations presents a series of difficulties. The costs of enforcing C&C policies depend on the economic incentives towards non-compliance and on the complexity of the C&C policy. The latter is of special relevance in the case of water, since the status of the water resources is the result of the action of multiple agents, and this makes complex to command and control all the factors involved to achieve the desired objectives (including those to impose law-enforcement mechanisms which require structures to inspect and apply fines and penalties, with increasing difficulties because of the magnitude of the problem). This is particularly true when the system is operating under critical situations of water stress (multiple sources of point pollution, overexploitation, etc.) that increase the marginal value of water and strengthen the incentives towards non-compliance (Porto and Lobato, 2004). In addition, C&C policies are often non-flexible and lack adaptive capacity (Pahl-Wostl et al., 2010). This is due to the fact that C&C policies either ignore uncertainties or assess them quantitatively in a way that ignores the non-linear changes that are making surprise and crisis increasingly common in many Mediterranean river basins worldwide (e.g., climate change). Consequently, C&C policies may be effective only for roughly stable systems with reliably recurring phenomena.

The relevant shortcomings associated with the implementation of C&C policies in combination with supply policies have made researchers to question the adequacy of the current paradigm[14] for water management. In fact, some have argued in favor of a looser water management approach that does not aspire to comprehensive

[13] Although in practice this norm has been seldom enforced (WWF, 2006).

[14] By paradigm we refer to the "intellectual and operational environment within which scientists 'do' science. It shapes the nature of problems to be addressed as well as the methods to be used and the interpretive lens through which the legitimacy and utility of findings are judged" (Pahl-Wostl et al., 2010).

command and control (Bucknall, 2006; Mayntz, 1998; Pahl-Wostl et al., 2010). In any case, this does not mean that C&C policies should be removed from the water policy mix; rather the contrary, both supply policies and economic incentives are inherently linked to C&C policies, which are necessary in order to define a common legal framework in which agents interact and to set a general direction in water policy for the achievement of collectively agreed goals. This is the case of Drought Management Plans (DMPs): although *per se* and even in combination with supply policies DMPs may be unable to attain the objectives of the WFD, they are a prerequisite for the development of efficient and effective water markets and drought insurance systems that may lead to a more sustainable water use.

DMPs are inspired in the drought contingency plans implemented in the US since the ‘80s and thus follow similar rules (NDMC, 2013). Basically, DMPs define the precise thresholds of possible drought situations and set the water constraints that will come into force in each of these cases, with the aim of guaranteeing water supply to priority uses. Drought thresholds are obtained from the historical assessment of water supply, while the extent of water constraints varies from one basin to another and depends largely on the ratio between water demand and water supply, being more restrictive in overexploited basins and focusing on agricultural uses (the water use with the lowest priority) (EC, 2008). As a result, the declaration of a drought will automatically reduce, in a predictable amount, the quantity of water delivered to the irrigation system from publicly controlled water sources.

In spite of being relatively new and voluntary, DMPs have rapidly spread across Southern EU countries such as France, Italy, Portugal and Spain[15]. In particular, Spain has pioneered the adoption of DMPs and currently every inter-regional (NUTS2) river basin in the country has already approved its DMP. This is particularly shocking if we consider that there are no assessments available on the potential impact of DMPs over the environment and over the productive activities exposed to water restrictions. Since DMPs focus exclusively on surface water and do not develop any instrument to regain control over loosely controlled and overexploited aquifers, we may expect that during droughts the pressures over surface water are at least partially transferred to groundwater through illegal abstractions. This effect

[15] Unlike other water management instruments such as River Basin Management Plans, DMPs are not prescriptive. However, apart from these set of Southern EU countries, DMPs have been also implemented in Finland, the Netherlands and the UK.

was not considered in the development of DMPs and its impact is basically unknown.

The following paper, prepared by the doctoral candidate and the Prof. Carlos Mario Gómez Gómez, intends to help bridge this gap. The paper develops a stochastic methodology to estimate the expected water availability and the potential for illegal groundwater abstractions in agriculture resulting from the decision rules of the recently approved DMPs in Spain. This method is illustrated with an application to the Segura River Basin. The paper is entitled *Do drought management plans reduce drought risk? A risk assessment model for a Mediterranean river basin*, and was published in Ecological Economics in the year 2012. Different versions of this paper were presented in the *86th Annual Conference of the Agricultural Economics Society* in Warwick, UK (16-18 April 2012) and in the *International Society of Ecological Economics 2012 Conference – Ecological Economics and Rio+20* in Rio de Janeiro, Brazil (16-19 June 2012).

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Analysis

Do drought management plans reduce drought risk? A risk assessment model for a Mediterranean river basin ☆

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ABSTRACT

Groundwater resources are traditionally overexploited in arid and drought-prone regions with profitable irrigated agriculture, and the depletion of this groundwater results from a combination of the physical scarcity of surface sources and the lack of effective control of use rights on the part of water authorities. This is the case in the Segura River Basin of southern Spain. As a result, drought risks and structural deficits have steadily increased over the last 50 years. The Drought Management Plan recently approved by the Segura River Basin Authority aims to enforce more stringent water supply restrictions from surface sources, but the plan does not include any explicit policy to handle illegal groundwater abstraction. By using a stochastic risk assessment model, this paper shows that the implementation of the drought plan will increase the expected irrigation deficits of surface water and can, paradoxically, lead to higher drought and aquifer depletion risks than the traditional rules that the new plan replaces.

1. Introduction

Many semiarid and drought-prone regions have significant competitive advantages for irrigated agriculture because the land is abundant and cheap and few alternative uses for the land exist. Furthermore, solar radiation is guaranteed and, apart from the abundance of cheap labour, many of these areas are located near high-demand markets. In fact, everything except water seems to be in place for developing a prosperous agricultural sector. In this context, water for irrigation can become the critical production factor that determines the viability of, and returns from, the agricultural sector. This is the case in many European Mediterranean regions where the survival of a competitive and highly productive agriculture critically depends on the ability to satisfy the water demands of a water-intensive irrigation system.

In these regions, although the water demand for irrigation is high, water property rights are poorly defined and enforced. Therefore, during frequent droughts, incentives are in place to use more water than the amount provided from publicly controlled sources. In fact, when current demands cannot be handled by publicly controlled sources, farmers have powerful incentives to switch to the more dependable, mostly uncontrolled groundwater sources. Uncertainty, coupled with the legacies of past management actions, often leaves decision makers few options other than to reinforce the current trajectory of the system (Anderies et al., 2006). The resulting overexploitation of the aquifers may reduce the robustness and resiliency of the system and its ability to cope with future droughts, thus leading to a vicious circle of increasing risk, vulnerability and water scarcity (Anderies, 2005; Anderies et al., 2004; Anderies et al., 2006; Holling, 1973; Perrings, 1989; Ruttan, 2002).

Some important measures have recently been taken to tackle the structural problem of recurrent droughts in the European Union. In what was perceived as an advanced replacement of past emergency responses by the apparently more appropriate planned and anticipated risk management response, several river basin authorities from Spain, the UK, Portugal, the Netherlands and Belgium have recently approved their respective Drought Management Plans (DMP) (EC, 2008). Basically, for the case of drought events, these plans establish more stringent constraints to access to publicly provided water while guaranteeing priority uses, such as drinking water, and ensuring minimum environmental services. As a result, the declaration of a drought will automatically reduce, in a predictable amount, the quantity of water delivered to the irrigation system from publicly controlled water sources. The DMP defines the precise thresholds of possible drought situations and sets the water constraints that will come into force in each of these cases (EC, 2008). For example, in

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the Segura River Basin in Spain, a four-stage classification system is used (normality, pre-alert, alert and emergency). In the case of an emergency, an optimistic[2] 50% of planned irrigation resources will be conceded in an attempt to guarantee, as highest priority, the survival of ligneous crops (although water distribution can be revised by the local authorities). Less stringent water constraints are established for alert (75%) and pre-alert levels (90%) (CHS, 2010b).

The plans reduce, *de jure*, the water supply during drought events. However, neither the DMP nor the water authorities introduce any instrument to handle the illegal abstraction of groundwater.[3] The illegal abstraction of groundwater is a major cause of the increased scarcity of water and drought risk in arid and semiarid catchments and also represents an important limit to the ability of the water authority to reduce water use during droughts. In fact, the imperfect enforcement of property rights over groundwater use in several European Mediterranean basins raises some serious doubts about the effectiveness of the DMP. Reductions in water supply from controlled sources, although proven efficient regarding surface water, are more difficult to enforce regarding legal and illegal groundwater sources (CHS, 2010a; Llamas, 2007). As has happened in the past, farmers may attempt to use informal and more reliable groundwater to compensate for the lack of formal surface water. Under the existing drought management rules, aquifers can be considered an insurance against drought,[4] making drought risk equivalent to groundwater depletion risk.

Controlling property rights is a necessary condition for managing the collective challenge of water scarcity and drought risk. The main hypothesis in this paper is that when water property rights are not perfectly enforced, making the formal water supply contingent on drought levels can paradoxically worsen both the water deficits and the risk of drought. To test this hypothesis, we develop a methodology to compare and assess the water supply deficits resulting from two alternative drought responses. In the first case, the baseline response results from the traditional decision rules historically applied in the basin. In the second case, the counterfactual response stems from the decision rules of the recently approved DMP.[5]

The basic conclusion of this paper is that if the new decision rules are not complemented by proper enforcement of water use rights, they will lead to increased water deficits and will reinforce the existing incentives to increase the depletion of the largely uncontrolled groundwater resources

The paper is structured as follows. In Section 2, we introduce the area where the case study is applied, the Campo de Cartagena agricultural district in the Segura River Basin (Spain). Section 3 presents the risk assessment model, and Section 4 presents and discusses the results obtained under the two alternative decision rules. Section 5 concludes the paper.

2. Background for the Case Study: Campo de Cartagena, Segura River Basin (Spain)

Because most of the variables involved are site- or crop-specific, such as rainfall, water demand, water supply and risk exposure, we illustrate each step of the model with the results for the particular case of the ligneous crops in the Campo de Cartagena agricultural district in the Segura River Basin (SRB).[6]

The SRB is a semi-arid water scarce basin exposed to an increased drought risk, and it is characterised by an imperfect enforcement of water use rights. For example, in 2008, according to the rainfall-runoff models used by the water authority, the average household, manufacturing industry and agriculture demand was estimated to be 1.9 billion cubic metres per year (1900 million cubic metres or hm^3, 85% of which was from irrigated agriculture), whereas the average renewable resources amounted to only 0.75 billion cubic metres (CHS, 2010a, 2011). These data thus yield a water exploitation index greater than 2.5.[7] Indeed, apart from the transfer of water from the Tagus River Basin, which has never accounted for more that 20% of the Segura water demand,[8] strong evidence (CHS, 2010a, 2011; WWF, 2006) suggests that the existing water supply deficit of the last several decades has been effectively covered by using the mostly uncontrolled groundwater sources.[9] [10] Rather than enforcing property rights by closing illegal mills, the traditional response has been to tolerate offenders[11] (CHS, 2010a; Llamas, 2007). Not surprisingly, the drought risk has increased along with the increase in water scarcity, and as the evidence presented in this paper shows, under the current water supply and demand, a drought can occur in one of every six years.

Campo de Cartagena, in the SRB, is an agricultural district with approximately 13,000 ha of irrigated ligneous crops (28.9% of the total irrigated land), which demand approximately 58 million cubic metres (hm^3) of water for irrigation in a normal hydrological year, of which approximately 16.7 hm^3 per year come from already overexploited aquifers (CHS, 2010a; MARM, 2007). Although it suffers from severe water scarcity, Campo de Cartagena, where the main ligneous crop is citrus fruit (CHS, 2011), is one of the largest and most profitable irrigated areas in Spain (CHS, 2010a), with production levels well over 20,000 kg/ha for some fruit trees (such as lemon, mandarin, orange and peach trees) (Pérez et al., 2011). Thus, the incentives for aquifer overexploitation are high, even in the presence of high abstraction costs.

The three aquifers in the Campo de Cartagena agricultural district, Carrascoy, Victorias and Campo de Cartagena, are overexploited even in non-drought periods. In a normal hydrological year, irrigation resources from these aquifers account for 29% of the irrigation demand,

[2] During past drought events, the conceded observed irrigation resources have reached, in many cases, levels well under the 50% of the initially planned irrigation resources. This was the case in the last drought in 2005–2008, when the conceded irrigation resources were less than 25% of the resources initially planned for the entire period (CHS, 2010b and 2011).

[3] On the contrary, river basin authorities have explicitly postponed compliance with the environmental European quality and state standards for aquifers beyond the initially planned deadline of 2015 (CHS, 2010b, 2011; EC, 2003)

[4] The traditional response against illegal water abstraction has been the result of infrastructure and the concession of additional irrigation rights (Gómez, 2009). This partly explains why irrigated land in CHS has grown more than 275% since 1990.

[5] No drought has been declared since its implementation; however, this is the result of a succession of relatively rain-abundant years (CHS, 2011).

[6] Campo de Cartagena is, simultaneously, one of the most overexploited and profitable agricultural districts in Spain (CHS, 2010a).

[7] The *water exploitation index* (WEI) is calculated as the ratio of total freshwater abstraction over total renewable resources. According to the European Environment Agency (2009), this index was 1.27 in 2003, indicating a meaningful trend towards a greater water scarcity in the last 20 years. Previous studies (Martínez Fernández and Esteve Selma, 2002) estimated that water consumption was already 2.25 times greater than the available renewable resources nearly a decade ago.

[8] Tajo-Segura Water Transfer was intended to provide an average of 55% of the total water resources in SRB and 35.78% of the irrigation resources between 2005 and 2010 (CHS, 2010b).

[9] The SRB accumulated groundwater overexploitation amounts to 7000 hm^3 (CHS, 2010b), including aquifers whose resources have been exhausted to such a degree that, even in the absence of more abstractions, it would take more than a century for them to completely recover. This is the case with the Alcoy-Sopalmo aquifer, where during some hydrological years, it has pumped out twenty times its renewable resources (CHS, 2010b).

[10] This occurs even though the granting of new concessions in the Segura River Basin has been forbidden since 1986 because of the significant water scarcity. Nevertheless, irrigated areas increased between 1990 and 2000 at a rate of 6500 ha/year (MMA, 2005). For example, in Campo de Dalias (Almería), the number of hectares farmed under plastic has tripled from 1980 to 2005, even though the drilling of new wells is prohibited. In the Segura River Basin, approximately 100,000 ha was irrigated with water illegally abstracted in 2005 (IDRUICM, 2005). See WWF (2006).

[11] The concession of new water use rights has been legally forbidden in the Segura River Basin since 2005, when aquifers were declared overexploited. Nevertheless, agricultural use increased by 5% each year since 2005 (CHS, 2010a, 2011). This is possible because of a lack of control over irrigation water demand. For example, only 155,313 ha of the 225,356 ha irrigated in Murcia (71.4% of the total irrigated land in the SRB) are officially registered by the water authority.

36% of which is non-renewable groundwater (CHS, 2010a). This over-exploitation is further exacerbated by the low technical efficiency of the abstraction, distribution and irrigation systems (25.5% according to CHS, 2010b) because only one-fourth of the water abstracted effectively contributes to satisfying the agronomic water requirements.

3. The Risk Assessment Model

To analyse the alternative drought management rules, we use a two-stage risk assessment method. The first stage consists of representing both the water required and the water available for a given set of ligneous crops at any moment in time as stochastic variables. In the second stage, we use these stochastic variables to determine the resulting water supply deficit associated with each decision rule. We can describe the two stages as follows:

- The first stage uses a standard method to obtain water requirements for each ligneous crop. We compare the evapotranspiration requirements with the amount of water available, which is from the following five sources: three stochastic sources (rainwater, runoff and stored water), the existing stock of groundwater and a variable but deterministic amount of non-conventional sources (wastewater reuse and desalinated water).
- The second stage allows us to determine the amount of water delivered to the irrigation system in accordance with the two alternative decision rules (traditional vs. drought contingency rules) and serves to measure the resulting excess demand for water as well as the moral hazard incentive to engage in illegal abstractions. The alternative decisions are obtained as follows:

i) In the baseline (traditional) case, the water authority decides the amount of surface water to be delivered to the irrigation system using the same discretionary rules that can be deduced from past decisions, which basically depend on the amount of runoff observed in any moment in time.
ii) In the alternative case, the water authority follows the decision rule approved as part of the DMP. When the natural supply of water is "normal", that is, the stored water and/or runoff may be sufficient, the decision is the same as in the traditional rule. However, in the case of a drought emergency, that is, in an alert or pre-alert state (which occur with a probability of 14% in our model), the amount of water delivered must be adjusted to the specific predetermined thresholds.

3.1. First Stage. The Decision Context: Water Requirements and Water Availability

Following the Spanish Ministry of Environment standard method (MARM, 2009b),[12] the amount of water required by a single crop, or its evapotranspiration (ET), is measured by using the evapotranspiration registered during the period from 1941 to 2009 (MARM, 2009b). In the case of irrigated crops, these water requirements are partially covered by the effective rainfall (ER) received from nature, which is a function of rainfall (a stochastic variable in the model). Thus, the amount of water required from the irrigation system, or the agronomic water required (WR) by a particular crop, is equivalent to the difference between the crop's evapotranspiration (ET) and the effective rainfall (ER). Agronomic water requirements can either be satisfied or not satisfied, depending on the region's natural capital (stochastic runoff) and human capital (surface water stored).

The effective coverage of the agronomic water requirements depends on three stochastic variables: rainfall, runoff and surface water stored. We consider the probability density function (PDF) of these three factors to determine the water supply at any moment in time.

Table 1
Rainfall Gamma function. The dependent variable is mm of rainfall.

Variable	Coefficient
a (Scale)	16.358[a]
	(2.821)
b (Shape)	22.9964[a]
	(2.286)
No. of observations	68

Source: Authors' elaboration from MARM, 2009b.
[a] Significant at the 1% level.

3.1.1. Effective Rainfall

Effective rainfall (ER) is the amount of rainfall in mm (p) that effectively contributes to satisfy evapotranspiration[13]:

$$ER = g(p). \qquad (1)$$

To represent ER_i under every possible state of nature, the observed data were adjusted to a probability density function (PDF)[14] that allows assigning a probability ($y = h(p)$) to each rainfall level (p). This function is obtained as the best fit gamma function[15] of the following type (Martin et al., 2001; McWorther et al., 1966):

$$y = z(p|a,b) = \frac{1}{b^a \Gamma(a)} p^{a-1} \exp\left(\frac{-p}{b}\right) \qquad (2)$$

where a and b are, respectively, the scale and the shape parameters. Table 1 presents the maximum likelihood estimators (MLEs) of this function's parameters. As Fig. 1 indicates, higher probabilities correspond to rainfall levels that are low or even very low for a region supporting a highly productive and water-dependent agriculture.[16]

The water deficit (WR) representing the part of evapotranspiration (ET) that is not covered by effective rainfall (ER) is also a stochastic variable, which can be defined as:

$$WR = ET - g(p). \qquad (3)$$

3.1.2. Runoff

The amount of water available to cover the agronomic water requirements is estimated using two proxy variables measured in percentage units. The first proxy variable is the percentage of annual cumulative runoff over the river basin surface water storage capacity (r), and the second proxy variable is the percentage of water stored over the river basin surface water storage capacity at the beginning of the crop season (s) (CHS, 2010b; Gómez-Ramos et al., 2002). Both are stochastic variables.

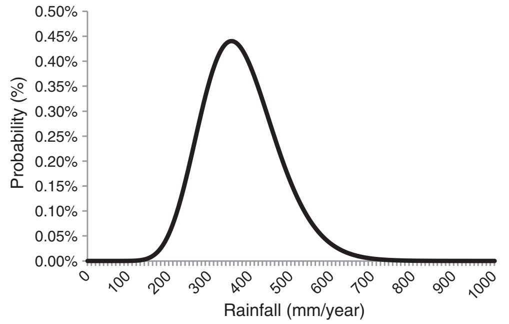
[12] MARM methodology follows a combination of the Thornthwaite and Penman-Monteith Methods (see, for example, Allen et al., 2006).

[13] Effective rainfall (ER) is estimated using the Soil Conservation Service–USDA methodology for Spain (Cuenca, 1989), and it is a function of humidity deficit (f(D)), rainfall (p) and evapotranspiration (ET). It is measured in annual mm: $ER = f(D) \cdot [1{,}25\ p^{0,824} - 2{,}93] - 10^{0,000955 \cdot ET}$.

[14] Data on cumulative annual rainfall are obtained from the *Sistema Integrado de Información del Agua* (SIA) (MARM, 2009b) for the period 1941 to 2009.

[15] The gamma function is defined by a scale parameter (a) and a shape parameter (b). It is consistent with rainfall measures because negative values are not allowed. The function reaches a maximum for intermediate values, decreases according to its scale parameter and converges to a normal distribution function as the shape parameter increases.

[16] The Segura River Basin (SRB) is exposed to a higher meteorological drought risk in Spain. The average evapotranspiration is similar to that of the Guadalquivir River Basin, although the time distribution is concentrated in low values (90% of rainfall values are between 400 and 800 mm, whereas, for example, values are above 500 mm with a 92% probability in the Ebro River Basin).



Source: Authors' elaboration from MARM, 2009b. See table 1.

Fig. 1. Rainfall probability density function, SRB, 1941–2008.Source: Authors' elaboration from MARM, 2009b. See Table 1.

Following Martin et al. (2001), we adjust the runoff probability distribution function to a gamma function.[17] This allows assigning a probability (q) to each runoff level (r):

$$q = f(r|a,b) = \frac{1}{b^a \Gamma(a)} r^{a-1} \exp\left(\frac{-r}{b}\right) \tag{4}$$

Table 2 and Fig. 2 show the best fit parameters for the runoff function.

3.1.3. Available Surface Stored Water

Following Gómez-Ramos et al. (2002) and Pérez et al. (2011), we adjust the probability distribution function of the level of available stored surface water by using the Weibull function,[18] which allows assigning a probability (w) to each stored water level (s)[19] (see Table 3 and Fig. 3):

$$w = j(s|a,b) = \frac{b}{a}\left(\frac{a}{b}\right)^{b-1} \exp\left(-\left(\frac{s}{a}\right)^{b}\right). \tag{5}$$

3.2. Decision Rules

At the beginning of each crop season, the water authority observes the level of water stored in the reservoirs and assesses the overall irrigation water required (*TIR*).[20] Accordingly, the water authority then applies a rule to determine the amount of water to be delivered to the crop fields.[21] The amount of irrigation resources actually delivered each year is a public decision that is based on water availability, and it may consist of using traditional decision rules (baseline) or applying the new decision rules of the recently approved DMP. We now analyse separately the two types of decisions (Fig. 3).

3.2.1. Traditional Decision Rules to Determine Water Delivery for Irrigation

In contrast with the situation created by the recently approved drought plans, the decision rules followed thus far have been the result of a combination of social agreements, opinions of expert judges and discretion with no written rules to be applied in any case, depending on the water available for the crop season. To formalise these decisions, we use the available data on the amount of water effectively delivered to farmers measured as a percentage of irrigation resources conceded over TIR. Available data span a range of 15 years (1992 to 2007) (CHS, 2010b), and as is normal in this type of analysis, the number of observations is fewer than required by a robust estimation of a probability distribution function. To compensate for the problem caused by the small number of observations, we follow the standard approach of increasing the sample size by representing the percentage of TIR satisfied as a proportion of runoff, r[22] [23] ($h(r)$) by using ordinary least squares (Gómez-Ramos et al., 2002).[24] The function relating $h(r)$ with runoff is presented in Table 4.

Finally, the effective surface irrigation resources ($EIR(r)$), or the part of the irrigation resources (*TIR*) that effectively satisfy evapotranspiration, can now be expressed as a function of the runoff (through $g(h)$) and the overall efficiency of the irrigation system (e_s):

$$EIR(r) = TIR * h(r) * e_{sw}. \tag{6}$$

Other publicly controlled water sources, such as the groundwater legally used (gw), the treated water (tw) and the desalinated water (dw), are provided to farmers in proportion to the irrigation resources delivered ($h(r)$)[25] from reservoirs. The amount of water delivered from each of these sources is converted into an effective irrigation resource by using its own technical efficiency index (e_{gw} for groundwater, e_{tw} for treated water and e_{dw} for desalinated water),[26] as follows:

$$gw(r) = \frac{\lambda}{\eta} * \mathrm{TIR} * h(r) * e_{gw} \tag{7}$$

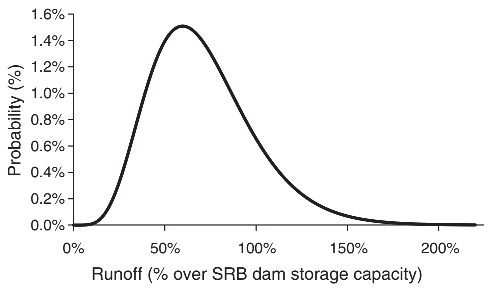
$$tw(r) = \frac{\gamma}{\eta} * \mathrm{TIR} * h(r) * e_{tw} \tag{8}$$

Table 2
Runoff gamma function. The dependent variable is the percentage of runoff over the total surface water storage capacity.

Variable	Coefficient
a (Scale)	6.1813[a]
	(1.088)
b (Shape)	0.1143[a]
	(0.012)
No. of observations	68

Estimated by maximum likelihood. Standard errors in parentheses.
Source: Authors' elaboration from MARM, 2009a.
[a] Significant at the 1% level.

[17] Runoff values range from 0% to 225% over the river basin dam storage capacity.
[18] The Weibull distribution is a continuous probability distribution with a scale parameter (a) and a shape parameter (b).
[19] The s data series, as a percentage of the total dam storage capacity, is obtained from *Anuario de Aforos* (MARM, 2009a).
[20] TIR is the maximum amount of irrigation resources that can be conceded in an ideal hydrological year. Spanish river basins estimate TIR as the agronomic water required to cover the 80th percentile of annual historical evapotranspiration (from 1941 to 2009) with a global efficiency of the water provisioning system of 60% (MARM, 2008). TIR is then higher than %TIR, and it is generally higher than WR.
[21] The irrigation resources actually conceded by the river authority in the SRB cover only a percentage of the estimated TIR (*%TIR*).
[22] The r data as a percentage of dam storage capacity were obtained from *Anuario de Aforos* (MARM, 2009a).
[23] Stored water (s) was not found to be statistically correlated with the percentage of TIR satisfied, which could be a consequence of the small storage capacity of the Segura River Basin. The ratio of reservoir storage capacity (1141 hm^3) over average yearly water use (1905 hm^3) is only 60% in the SRB, far lower than that of the drought-prone Guadalquivir (238%) and the rainfall-abundant Ebro River Basin (90%) (see: CHE, 2011; CHG, 2011; CHS, 2011).
[24] For values of *TIR* over 100%, the function is truncated and equals 1.
[25] In an average hydrological year, Campo de Cartagena irrigation resources come primarily from dam stored water (65.31%, η, 37.6 hm^3 of effective water) and groundwater (29%, 16.92 hm^3 of effective water, λ). Desalinated water (0.39%, θ) and treated water (5.3%, γ) are negligible (3.32 hm^3 of effective water) (MARM, 2007). These percentages are assumed to be constant in the model.
[26] Piping and irrigation techniques determine the final amount of effective water applied to satisfy a certain amount of a crop's water demand. Global efficiency of the system for the Campo de Cartagena region is approximately 87% for dam stored water, 60% for desalinated water and treated water and 25% for groundwater (CHS, 2010a; MARM, 2007).



Source: Authors' elaboration from MARM, 2008. See Table 2.

Fig. 2. Runoff probability density function, SRB, 1941–2008.Source: Authors' elaboration from MARM, 2009a. See Table 2.

Table 3
Surface water stored: Weibull function. The dependent variable is the percentage of dam stored water over dam storage capacity.

Variable	Coefficient
a (Scale)	0.3411[a] (0.063)
b (Shape)	4.1286[a] (0.497)
No. of observations	68

Estimated maximum likelihood. Standard errors in parentheses.
Source: Authors' elaboration from MARM, 2009a.
[a] Significant at the 1% level.

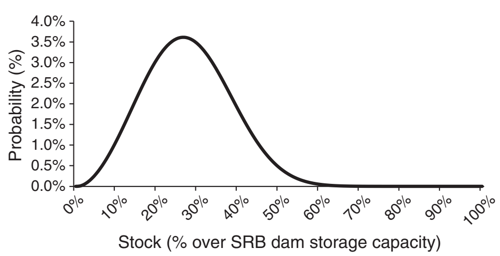
$$dw(r) = \frac{\theta}{\eta} * \mathrm{TIR} * h(r) * e_{dw} \tag{9}$$

The percentage of the evapotranspiration satisfied (*%ET*) can now be obtained from expressions (7) through (10), as follows:

$$\%ET_{p,r} = \frac{g(p) + EIR(r) + gw(r) + tw(r) + dw(r)}{ET}. \tag{10}$$

Each *%ET* has an associated probability ($prob_{\%ET}$), which depends on runoff (r) and rainfall (p) values. Using expressions [2] and [4], this probability can be expressed as follows:

$$prob_{\%ET_{p,r}} = f(r) * z(p). \tag{11}$$



Source: Authors' elaboration from MARM, 2008. See Table 3.

Fig. 3. Dam stored water probability density function, SRB, 1941–2008.Source: Authors' elaboration from MARM, 2009a. See Table 3.

Table 4
Irrigation resources estimation under the traditional decision. The dependent variable is a percentage of irrigation resources conceded in the SRB over TIR.

Variable	Coefficient
Runoff (percentage over dam storage capacity)	1.351[a] (.131)[b]
R2	89.14
Adjusted R2	88.31
No. of observations	15

Source: Authors' elaboration from CHS (2010b).
[a] Significant at the 1% level.
[b] Estimated by maximum likelihood. Standard errors in parentheses.

The expected level of evapotranspiration coverage ($E_{\%ET0}$) and the resulting expected irrigation deficit (*ID*) in the traditional rule scenario can be represented as:

$$E_{ET} = \int_{r=0}^{225} \int_{p=0}^{1300} [z(p) * g(p) + f(r) * (EIR(r) + gw(r) + tw(r) + dw(r))] \tag{12}$$

$$ID = ET - E_{ET}. \tag{13}$$

Illegal groundwater abstraction is a positive function of irrigation deficits. The use of surface water is observable and controlled by the water authority within the limits of the existing legal property rights. In contrast, access to groundwater is a moral hazard decision made by the farmer and is unobservable by the water authority. When water allowances from publicly controlled water resources fall short with respect to agronomic needs, as the evidence in the Segura River Basin shows, farmers will have positive incentives to seek uncontrolled groundwater sources. Illegal groundwater abstraction (*GW*) is then a positive function of the irrigation deficit:

$$GW = c\left(\frac{ID}{e_{gw}}\right). \tag{14}$$

3.2.2. DMP Decision Rules over Water for Irrigation

The recently approved DMP for the SRB quantifies the particular situation at hand and the severity of the problem by using an objective and publicly observable drought index, I_e. This plan establishes the following four drought thresholds (CHS, 2010b) i) when water stored levels are regarded as *normal* ($I_e > 0.5$), there are no additional explicit restrictions, and thus water delivery (*%TIR*) is the same as in the baseline or traditional rule scenario; ii) water for irrigation is reduced by 10% ($h = 0.9$) when available water falls below the pre-alert threshold ($0.35 < I_e \leq 0.5$); iii) if the alert limits are exceeded ($0.2 < I_e \leq 0.35$), water for irrigation is reduced by at least 25% ($h = 0.75$); and iv) in emergency situations ($I_e \leq 0.2$), water for irrigation is halved ($h = 0.5$). According to historical data, a drought is quite likely in the SRB, occurring with a probability of 14%.[27]

In the case of Campo de Cartagena in the SRB, the drought index (I_e) depends on the observed values of both runoff and stock[28]

[27] This is a minimum threshold. Historical data underestimate drought risk because the data do not consider that today's water resources are jeopardised significantly more than in the past.

[28] I_e is calculated as follows (CHS, 2010b):

$$I_e = \frac{1}{2}\left(1 + \frac{V_i - V_{med}}{V_{max} - V_{min}}\right), \text{ if } V_i \geq V_{med}$$
$$I_e = \frac{1}{2}\left(\frac{V_i - V_{min}}{V_{med} - V_{min}}\right), \text{ if } V_i < V_{med}$$

where V_i is an indicator that is unique for each *junta de explotación* (a group of agricultural districts of comarcas). In *Sistema Cuenca*, which is Campo de Cartagena's corresponding sub-basin, V_i, is obtained as follows: $V_i = \frac{2 * DSC * r + DSC * s}{3}$ Where r is the runoff as a percentage of the total dam storage capacity (*DSC*) and s is dam stored water as a percentage of the total *DSC*. Using r and s maximum, minimum and average observed values during the reference period, we obtain V_{max}, V_{min} and V_{med}, respectively

(CHS, 2010b). Therefore, we define $l_{r,s}$ as a discrete water restriction variable whose value depends on the drought index (and thus on runoff and stock values) and its corresponding h. As the empirical data suggest, the estimated satisfied agronomic crop requirements under the new drought plan are too optimistic compared with past events. Therefore, we set $l_{r,s}$ as the minimum between $h(r)$ defined in the baseline scenario and the SRB's DMP parameters above (h):

$$l_{r,s} = \begin{cases} min(h(r), 0.5), \text{ if } I_e \le 0.2 \\ min(h(r), 0.75), \text{ if } 0.2 < I_e \le 0.35 \\ min(h(r), 0.9), \text{ if } 0.35 < I_e \le 0.5 \\ h(r), \text{ if } I_e > 0.5 \end{cases}. \tag{15}$$

In every case, the percentage of evapotranspiration satisfied with rainfall and surface water ($\%ET2$) and its associated probability ($prob_{\%ET2}$) can be obtained from:

$$\%ET2_{r,s,p} = \frac{g(p) + \frac{l_{r,s}}{h(r)} * (EIR(r) + gw(r) + tw(r) + dw(r))}{ET} \tag{16}$$

$$prob_{\%ET2_{r,s,p}} = f(r) * z(p) * j(s). \tag{17}$$

We can also obtain expected evapotranspiration satisfaction and expected deficit under the Drought Management Plan scenario by conditioning evapotranspiration satisfaction to the impact of the drought threshold indices ($l_{r,s}$):

$$E_{ET} = \int_{r=0}^{225}\int_{p=0}^{1300}\int_{s=0}^{100}\left[z(p) * g(p) + f(r) * \frac{l_{r,s}}{h(r)} * (EIR(r) + gw(r) + tw(r) + dw(r))\right] \tag{18}$$

$$ID = \mathrm{ET} - E_{\mathrm{ET}}. \tag{19}$$

Again, illegal groundwater abstraction would be a positive function of irrigation resources [14]. The entire methodology must be performed for every crop in every agricultural district considered.[29]

4. Drought Decision Rules and Water Deficits

Table 5 compares the outcome of the two decision frameworks in terms of the expected rates of evapotranspiration covered and the associated irrigation deficits (in both volume and per cent units). The last row, *PotGW*, shows the expected amount of non-authorised water abstractions that would be required to fully cover the irrigation deficits in the Campo de Cartagena region with the existing technical efficiency of the irrigation system.

In the baseline, droughts occur with a 14% probability, and the expected deficit amounts to 1.82 hm^3 of effective water (this deficit is confirmed by the water authority in CHS, 2008), which means that the technical efficiency of the irrigation system would require the abstraction of an additional 7.28 hm^3. Implementing the decision rules of the drought plan will increase this expected deficit by 35% to 2.4 hm^3. As a result,[30] the implementation of the new planned decisions for drought conditions will add pressure to the already overexploited aquifers in the area because at least a portion of the increased supply deficit will be satisfied by increasing uncontrolled groundwater.

Water deficits and incentives for aquifer overexploitation are particularly high during drought emergency events ($I_{e_{t,j}} \le 0.2$), which occur one in ten years (with a probability of 9.88% in our model). A drought emergency would imply a severe cut in water allowances for irrigation, which would have a significant impact on evapotranspiration satisfaction and the irrigation deficit, decreasing production and income (Pérez et al., 2011). Thus, incentives for illegal groundwater abstraction, in this case, are even greater, as shown in the table below.

Table 5
Expected evapotranspiration satisfaction, expected irrigation deficit and expected potential illegal groundwater abstraction in absolute terms (hm^3) and as a percentage of ET satisfied (%**ET**) for all possible states of nature in the Campo de Cartagena agricultural district.

		Baseline scenario	DMP scenario	Difference
Total expected evapotranspiration satisfaction	E_{ET} (hm^3)	43.89	43.31	−0.59
	$E_{\%ET}$	94.73%	92.32%	
Expected irrigation deficit	ID (hm^3)	1.82	2.41	0.59
	$ID_{\%ET}$	3.99%	7.68%	
Expected potential groundwater depletion	$PotGW$ (hm^3)	7.15	9.45	2.3

Source: Authors' elaboration.

Compared with previous decision rules, the expected irrigation deficit will increase from 17% to 22% (see Table 6), and it will require an additional 8 hm^3 of water, meaning that there will be higher incentives to use poorly controlled groundwater sources. By trying to reduce water use, responses planned for drought conditions that are similar to those in the case of the Segura River Basin can reinforce the existing moral hazard incentives to groundwater depletion and thereby lead to the paradoxical result of decreased resilience and increased drought risk in the future.

5. Discussion and Conclusions

The results presented in this paper provide relevant insights not only within the field of ecological economics but also in the broader area of drought risk management. The main conclusion is that DMPs must be properly designed and should consider all possible water sources to guarantee that a comprehensive social–ecological water conservation framework is put into place. Otherwise, the water demand stemming from the implementation may result in local overexploitation of illegal water sources such as aquifers and thus result in a loss of resilience and robustness.

Such is the case in the SRB in Spain. Irrigated agriculture in this area is among the most extensive and most profitable in Spain (CHS, 2010a; Pérez et al., 2011), although its sustainability is compromised by structural water scarcity and recurring droughts (CHS, 2010a, 2010b; EEA, 2009). The farmers' traditional response to use groundwater as an insurance against drought (Llamas, 2007; WWF, 2006) generates a vicious cycle of higher water deficits, lower resilience and more frequent and severe droughts. This dynamic can be reversed only when water use is curved downward to match the long-term renewable resources of the river basin, which is not

Table 6
Expected evapotranspiration satisfaction, expected irrigation deficit and expected potential illegal groundwater abstraction in absolute terms (hm^3) and as a percentage of ET satisfied (%**ET**) under emergency in the Campo de Cartagena agricultural district.

		Baseline scenario	DMP scenario	Difference
Total expected	E_{ET} (hm^3)	37.84	35.82	−2.02
Evapotranspiration satisfaction	$E_{\%ET}$	82.76%	78.34%	
Expected irrigation deficit	ID (hm^3)	7.88	9.90	2.02
	$ID_{\%ET}$	17.24%	21.66%	
Expected potential groundwater depletion	$PotGW$ (hm^3)	30.90	38.83	7.92

Source: Authors' elaboration.

[29] Most parameters in the model can be taken at a river basin level, except K coefficients and system global efficiency, which are unique for each crop and district, respectively.

[30] Only 12 proceedings for illegal water abstraction have been initiated between 1996 and 2005 in the SRB, which offers a perspective on the immunity under which offenders operate (WWF, 2006).

possible without the enforcement of existing water property rights (Raffensperger, 2011). The existing data and the risk assessment analysis presented in this paper suggest that more stringent water constraints over publicly controlled water sources, which are the essence of the recently approved Drought Management Plans, will not effectively reduce drought risk. Furthermore, without recovering the control of groundwater resources, these norms will only contribute to the water scarcity, making water an even more valuable resource and resulting in new incentives for farmers to engage in the moral hazard type of behaviour that now pervades irrigated agriculture in many Mediterranean areas including the SRB.

For example, under the new Drought Management Plan in the SRB, a likely drought with a rainfall less than 400 mm and a drought index below 0.2 would lead to an expected deficit in effective irrigation water of 18.23 hm^3, thereby requiring the abstraction of as much as 71.51 hm^3 of groundwater (more than four times the amount required in a normal hydrological year). This event actually occurred between 2005 and 2008, when the drought index remained below the emergency level throughout the majority of the period. The failure of the emergency responses used at that time was one of the principal arguments for designing the drought plans that were approved in 2008. However, the emerging decision rules ignore the basic fact that quantitative water constraints can be successful only if water property rights are properly designed and enforced. In effect, according to the results presented in this paper, the Drought Management Plan will make future droughts more likely and more severe.

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Part III

Economic Instruments for Water Management

5 Economic Instruments for water management

The current water crisis is now recognized as being largely a crisis of governance[16], and not of resources or technological problems (Bucknall, 2006). In fact, while the technical capacity of the society to put additional amounts of water at the service of growing water demand has increased exponentially, society seems to have failed to acknowledge nature's physical constraints. Consequently, whereas science has developed technical water management to a very large extent, considerations of social, political, institutional and financial order (i.e., economics) are still treated in an incipient form, with major problems persisting. This failure is perfectly exemplified by the lack or the inadequacy of the current policy mix to match the decisions made by the different water users in the local economy with the ability of the existing water resources to satisfy these uses in a sustainable manner.

In order to overcome the current water crisis, some researchers and policy makers have demanded a paradigm shift in which conventional water policy is complemented with economic instruments in order to replace the traditional notions of control and government-led planning by those of incentives, motivation and multi-level governance (Pahl-Wostl et al., 2010). However, evidence demonstrates that there is still a major gap between the political rhetoric and the operational level. With the exception of water markets and water pricing, economic instruments for water management are seldom found outside the academia, and in many cases they consist of financial tools disguised as economic instruments to make the ultimate goal of raising revenues more acceptable (Strosser et al., 2013).

As a result of the little evidence available and of the misuse of economic instruments, there is significant confusion regarding what an economic instrument for water management actually is. Although it is generally accepted that taxes, fees, subsidies and markets can be all considered economic instruments, there is still disagreement regarding the inclusion of non-market mechanisms in this group and the purpose and design of economic instruments. This confusion is perceivable

[16] Noteworthy, some of these governance failures have an economic explanation (Hanemann, 2006).

even in the academia. Different definitions have been made available, and in some points they conflict with each other. For example, NCEE (2001) considers that economic instruments are financial tools (i.e., market based) that “provide monetary and near-monetary rewards” for accomplishing environmental goals. In the same line, Stavins (2003) puts economic instruments at the same level as market based instruments and labels them as “harnessing market forces”, because if they are well designed and implemented, they “encourage agents to undertake pollution control efforts that are in their own interest and collectively meet policy goals”. Kraemer et al. (2003) provide a clearer definition and finally open up the category to non-market economic instruments, but they do not separate economic instruments for water management from revenue raising tools: “When the primary aim of an environmental charge or tax is not to create incentives but to raise revenue, the relevant distinction lies whether the revenue is earmarked or simply added to the general government budget”. Although financial instruments are of paramount importance for the accomplishment of the goals defined in water policy, they are not economic instruments for water management *per se*, since they do not need to pursue an environmental objective. In addition, even if financial instruments are earmarked for environmental purposes, the relevant economic instrument here would be the mechanism in which this money is used, and not the revenue raising tool. ONEMA (2009) provides a very similar definition and incurs in the same mistake as Kraemer et al. (2003).

In spite of the contradictions already mentioned, all the definitions above agree to point out that the key elements in an economic instrument are those of incentives, motivation and voluntary choice. Moreover, they all stress that at least one of their objectives should be that of adapting individual decisions to collectively agreed environmental goals. Strosser et al. (2013) gather up all these contributions and create a synthetic definition. According to these authors, economic instruments are “those incentives designed and implemented with the purpose of adapting individual decisions to collectively agreed goals (e.g. the environmental objectives of the WFD and of its “daughter” Directives)”. This implies that financial instruments aimed primarily at raising revenues are not economic instruments for water management (although cost-recovery can be a secondary objective of these); on the other hand, co-operative agreements and other non-market instruments that lead to behavioral changes may be economic instruments for water management even in the absence of financial transactions.

Economic instruments for water management are not a substitute of conventional C&C and supply policies; rather, they should be designed to complement them[17]. Nonetheless, once the potential of water works has been developed far beyond the capacity of the system and C&C policies have become more and more sophisticated without attaining the preset objectives of water policy, the alternatives available to achieve a sustainable water use must be found in a combination of new economic instruments capable of articulating the increasing water demand, limited water supply, water policy goals and the existing infrastructures and legal framework. This is exactly the situation in which we find many Mediterranean basins.

Economic instruments for water management have been consistently developed in Spain during the last decades. For example, water scarcity and recurrent droughts in Mediterranean basins (EEA, 2009) have promoted the adoption of water markets and have made Spain a laboratory to test the outcomes of this instrument under the particular EU legal framework[18]. Nonetheless, the experience of Spain with economic instruments is not restricted to water markets. Voluntary agreements (Gómez et al., 2013), subsidies (Lopez-Gunn et al., 2012), water pricing instruments (through water tariffs) (EEA&OECD, 2013) and risk management schemes (Pérez-Blanco and Gómez, 2013) have been also explored and/or implemented.

In the next sections we present a series of methodologies that have been used in the assessment and in some cases in the implementation of innovative economic instruments for water management in Spain. The methodologies used are *ex-ante* and grounded in economic theory. Since these methods are quantitative, they are illustrated with case studies that provide some insightful results.

Therefore, our objective in the following sections is to present some methods and stylized facts that can be used to assess the implementability of some of the most relevant economic instruments that have been advanced in Mediterranean basins, namely, voluntary agreements, drought insurance and water pricing.

[17] This makes even more challenging to single out the actual contribution of an economic instrument to the water policy goals: economic instruments are never implemented in isolation from other supply and C&C policies (nor should they); and many macro-economic and sectoral changes that influence behavior and water use also take place at the same time (Strosser et al., 2013).

[18] Water markets have been widely developed through different legal figures, namely, lease contracts, water exchange centers, purchases of land to use water and inter-basin temporary trading, but also through informal water trade (Albiac et al., 2006; Calatrava and Gómez-Ramos, 2009; Garrido and Calatrava, 2009; Rey et al., 2011).

5.1 Voluntary agreements

In some specific contexts, water policy can attain significant welfare gains through the promotion of voluntary agreements that generate (or merge) incentives for a win-win situation. Building cooperative agreements is only feasible when private interest is somehow compatible with the actual purposes of water policy, such as the recovery of some ecological potential of the river system. In these cases, rather than altering market dynamics or defining a new set of rules that agents have to comply with, the goal of water authorities should focus on creating an environment that is favorable for the development of this type of agreements.

Voluntary agreements cover a wide spectrum and can be used to improve the environmental status of water bodies in different ways. This may range from the voluntary compliance with a certification scheme that signals a good environmental performance (Bratrich and Truffer, 2001) to bilateral agreements involving private and public agents (Gómez et al., 2013). Although voluntary agreements have been sometimes rejected as an economic instrument (NCEE, 2001; Stavins, 2003), they should be fully considered as such (Kraemer et al., 2003; ONEMA, 2009; Strosser et al., 2013).

In Spain, one of the most significant and successful experiences with these instruments is the voluntary acceptance by the hydropower operator of the Mequinenza-Ribarroja-Flix Dam Complex to release flushing flows designed to improve the qualitative status of the Lower Ebro (northeastern Spain). This agreement was possible due to the coordinated efforts of the hydropower operator, the water authorities and the scientific community. The role of the scientific community largely consisted in the design of the standard hydrograph of the flushing flows and the economic model that served to estimate the private revenue foregone following a cost minimizing implementation of flushing flows. Therefore, two models were used in this interdisciplinary research: i) first, a hydrological model (Batalla and Vericat, 2009) based on the sediment entrainment method (Kondolf and Wilcock, 1996); ii) second, an economic model (Gómez et al., 2013) aimed at minimizing the opportunity costs of flushing flows. The latter constitutes the core of this chapter.

The background for the implementation of this economic instrument is the construction of the large Mequinenza and Ribarroja dams in the 1960s, which significantly modified the hydrology of the Lower Ebro River (Batalla and Vericat, 2009). Although the river still experiences natural floods, its physical and environmental conditions have remarkably changed during the last decades (ERBA, 2008). These changes have resulted in local incision and riverbed armouring, re-vegetation of formerly active areas of the river channel, reduction of sediment inputs to the delta (modifying the dynamics of the estuary and resulting in salt intrusion) and, especially, the proliferation of macrophyte biomass (Batalla and Vericat, 2009; Vericat et al., 2006).

A macrophyte is an aquatic plant that grows in or near water. Although its proliferation is beneficial in lakes, where they are regarded as eco-indicators, in highly engineered rivers its presence evidences degradation, rather than good ecological status. In addition, macrophyte biomass may cause problems in water intakes and navigation, may constitute a threat to public health[19] and may hamper the regular functioning of irrigation pumping stations, hydropower and energy plants (Batalla and Vericat, 2009; ERBA, 2008). Prior to 2002, costly actions were adopted in order to mechanically remove macrophytes. At that time, the delivery of recurrent flushing flows in order to remove macrophyte biomass appeared as an alternative to avoid costly adaptation to degrading water conditions.

From a public perspective, the potential benefits of flushing flows were related to the partial recovery of the river regime, covering a wide range of benefits including the control of invasive species, the abatement of salt intrusion in the river mouth and the improvement of water quality along the river. However, the private interest of hydropower operators was in principle focused on removing the macrophyte biomass located close to the hydropower plants, which required a far less demanding flushing flow than one aimed at partially restoring the entire Lower Ebro down to the estuary. In any case, the hydropower operator proved to be willing to consider water flow patterns that were not only designed to maximize financial profits within the range of prevailing regulations, but also to deliver some improvements in the ecology of the river system. Besides, although at the beginning the possibility of compensation to the hydropower operator in exchange of the release of flushing flows was considered, finally the agreement was voluntary. This was largely owed to the low share of the hydropower operator’s annual income that the flushing flows represented (Gómez et al., 2013). In addition, the hydropower

[19] Macrophytes are also seen as the main cause of the summer plagues of black flies (*Simulium spp*.) which may transmit diseases such as *onchocerciasis* (river blindness).

operator could avoid the costly actions to mechanically remove macrophytes and convert the whole intervention into part of its corporate social responsibility strategy.

Since 2002 a series of flushing flows have been implemented. Initially, this was only for experimental purposes, supported by a research program to design flushing flows and to monitor and maximize their effectiveness. More recently, these efforts were integrated in the design of the river basin management plan and finished with the agreement to deliver two controlled flushing flows every year (in spring and autumn), deliberately defined to maximize macrophyte removal rates and implying the delivery of 36 million m^3 (hm^3) in 13 hours in each controlled flood (ERBA, 2008).

Despite the relative success of the voluntary agreement, recent evidence has shown that the effectiveness of flushing flows to restore the river is now lower than in the previous decade, even for macrophyte biomass removal (ERBA, 2013). Effects are better in the immediacy of big dams and hastily decrease with only marginal changes in the river estuary. Paradoxically, the success in improving the chemical status of the river in the last ten years seems to have increased the potential for macrophyte proliferation and boosted its rate of renewal after every controlled flood. Recent research indicates that flushing flows help in river restoration but are increasingly insufficient to offset the many hydromorphological changes affecting the Lower Ebro. Better designed environmental flows, presumably with a higher frequency and intensity, are required.

The following paper, prepared by the doctoral candidate, the Prof. Carlos Mario Gómez Gómez and the Prof. Ramon Batalla, presents an economic model that can be used to estimate the opportunity costs of the periodical release of flushing flows in rivers whose regimes are controlled by hydropower generating facilities. This economic model was designed to support decision making in the bargaining process among the different agents involved in the voluntary agreement in the Lower Ebro. The paper is entitled *Tradeoffs in river restoration: Flushing flows vs. hydropower generation in the Lower Ebro River, Spain*, and was published in the Journal of Hydrology in the year 2013.

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Tradeoffs in river restoration: Flushing flows vs. hydropower generation in the Lower Ebro River, Spain

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SUMMARY

Although the effectiveness of flushing floods in restoring basic environmental functions in highly engineered rivers has been extensively tested, the opportunity cost is still considered to represent an important limitation to putting these actions into practice. In this paper, we present a two-stage method for the assessment of the opportunity cost of the periodical release of flushing flows in the lower reaches of rivers with regimes that are basically controlled by a series of dams equipped with hydropower generation facilities. The methodology is applied to the Lower Ebro River in Spain. The results show that the cost of the reduced power generation resulting from the implementation of flushing floods is lower than the observed willingness to pay for river restoration programmes.

1. Introduction

Water is an economic asset necessary to sustaining life, the environment and the production of many valuable goods and services and should be managed accordingly. However, the prevailing paradigm considers water demand to be exogenous, and water policy, consequently, has traditionally focused on guaranteeing the supply of water services at affordable prices. As a result, during the last decades population growth and the improvement of living standards brought about by development have increased the pressures on water resources. The negative environmental effects stemming from this paradigm are visible for instance in the case of the European and North American rivers, where the need to satisfy a continuously growing demand for water and river services has resulted in increased water abstractions and polluted discharges along with gravel mining, canalisation, and successive modifications in river morphologies (e.g., Furse et al., 2006; Zawiejska and Wyzga, 2010; Batalla and Vericat, 2009).

Consequently, restoration of river ecosystems has become a priority for water management in the developed world, especially in the stressed lower reaches of its rivers (Gupta and Bravard, 2010; EC, 2000). However, restoration is often obtained at the cost of impairing the ability of water infrastructures to provide valuable socioeconomic goods and services, such as hydropower (Bednarek and Hart, 2005; Palmieri et al., 2001; Robinson and Uehlinger, 2003). There is thus a considerable interest in learning how to balance river restoration benefits with the production of goods and services provided by water infrastructures.

As a result of this interest, significant effort in scientific research has recently been mobilised in two important directions. Considerable progress has been made in the assessment of current ecological status and trends and in the design of effective technical alternatives to restoring some basic environmental functions of rivers. In particular, emerging research in biology and ecological engineering (e.g., Granata and Zika, 2007) shows that dams and other infrastructures that alter river systems can also be used as tools to reproduce artificially a portion of the functions performed in the past by the natural system. For instance, modifying the rules of hydropower dam operation to guarantee the periodic release of properly designed maintenance flows (namely, flushing flows) may effectively replace the role performed in the past by the natural floods characteristic of many rivers, which served to maintain the structure and functions of the river ecosystem (see Hueftle and Stevens, 2002; Vinson, 2001; Kondolf and Wilcock, 1996). Social sciences have also provided methods and results for the valuation of the economic and social benefits of potential improvements in the capacity of river systems to increase the quantity and range of those environmental services that might result from a successful restoration of river systems (such as

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recreation opportunities, biodiversity support, health services, water security and flood control) (see, for example, Hitzhusen, 2007; Turner et al., 2003 and Gupta and Bravard, 2010; CSIRO, 2012). However, there is still little research on the costs of practically applying the available options to improve rivers' ecology, which makes the opportunity cost of water the missing element for the assessment of the policy options at hand.

Information on opportunity costs plays a critical role in the evaluation of river restoration alternatives for a series of reasons: to find the most cost-effective way to improve the river environment and thus minimise the impact over marketable water services, to judge whether the associated cost is lower than the benefits expected from the improvement of the water environment (and to assess later whether the proposed measures are justified in the light of cost benefit criteria), to provide the critical information to assess what would, for example, be the minimum compensation demanded by water users for voluntarily adapting the use of the resource to certain new requirements and to know the real cost of harmonising the provision of water services and the improvement and protection of the water environment.

This paper aims to help bridge this information gap. The paper presents a model for the evaluation of the opportunity costs of implementing a given flushing flow programme in an area where the flow regime is driven by the operation of a hydropower facility. In such a situation, the requirement to release the flushing flow means that for certain precise periods of time, the outflow of water does not depend on the profit maximising criteria used by the hydropower plant (baseline scenario) but rather on an operating constraint imposed by an environmental authority (counterfactual flushing-flow scenario). The opportunity cost of such measures is therefore represented by the monetary losses of the concerned commercial activity, namely, hydropower. The overall question we want to answer can be presented as determining a financial value for the compensation required by a hydropower operator to voluntarily accept a predetermined programme of periodical artificial releases. The model is illustrated with an application to the Lower Ebro River, Spain.

2. The Lower Ebro River: river diagnosis and the need of flushing flows

The Lower Ebro River is located in the northeast of Spain and comprises the area located between the Mequinenza–Ribarroja–Flix Dam Complex (hereafter MRFDC) and the outlet of the river to the Mediterranean Sea (see Fig. 1). Water demand from agriculture is significant (1.200 million cubic meters/year, i.e., 90% of the total water demand), and runoff has been reduced by more than 20% as a result of increasing pressures from upstream and long-term changes in land use (i.e., afforestation). However, flows are still relatively abundant, and droughts are rare (ERBA, 2007). The main environmental concern in the area is related to the impoverished ecological status that resulted from the alteration of the river's hydrology and, subsequently, the channel morphology after the construction of the MRFDC (see Table 1).

The large Mequinenza and Ribarroja dams built in the 1960s substantially modified the flow regime of the Lower Ebro. Among other hydrological components, flood magnitude and frequency have been altered. Of particular interest for the river's ecological functioning is the 25% reduction, on average, of the relatively frequent floods (i.e., those with a return interval between 2 and 25 years) (Batalla et al., 2004). Although the river still experiences natural floods, and the impact of regulation is much smaller than that found in comparable large rivers, such as, for instance, the Sacramento and the San Joaquin Rivers in California (Kondolf and Batalla, 2005), the river's physical and environmental conditions have changed notably in the last decades (e.g., Batalla et al., 2006; Vericat and Batalla, 2006; Vericat et al., 2006; Batalla and Vericat, 2009). The main dam induced changes can be summarised as follows:

- Reduction of flood frequency and magnitude; floods provide the energy for maintaining an active river channel morphology, and this reduction has led to the loss of formerly sedimentary active areas, the encroachment of riparian vegetation and the narrowing of the channel.
- Reduction of the river's sediment load, which implies the erosion of the gravel fractions in the channel with no replacement from upstream and simultaneous riverbed armouring during small frequent floods and during high flow periods.
- Alteration of the river's ecology, as a compound effect of impoundment, exemplified by the low frequency of bed moving floods, slow moving waters, deficit of fine sediment, high temperatures and excess nutrient load. These combined alterations create a new functioning in the river ecosystem with consequences regarding the river's ability to provide key environmental services.

This new set of environmental conditions, together with similar changes in the upstream main tributaries, appears to explain the uncontrolled proliferation of macrophytes[1] in the Lower Ebro River channel (e.g., Goes, 2002; Palau et al., 2004). Macrophytes threaten river infrastructures, increasing operating costs, reducing the productivity of power-generating plants and water-pumping devices and reducing the ability of the river to provide navigation and recreation services. Competition for space and resources resulting from the stabilisation of dense macrophyte stands also affects the biology of the river ecosystem in many different ways. Macrophyte stands limit the access to microhabitats that are important for the growth and survival of juvenile fish, and the decomposition of growing organic matter depletes the water of its oxygen. Macrophytes communities also enhance flow resistance, thus exacerbating the reduction in flow velocity and trapping an important portion of fine sediment load (Batalla and Vericat, 2009).

Within this context, a considerable body of research has been devoted to the design and implementation of flushing flows as a means to improve the ecological status of the Lower Ebro River. These efforts started in 2002 following two notably dry years. These drought conditions encouraged cooperation between the hydropower operator, the water authorities and the scientific community. With the exception of two dry years in 2004 and 2005, flushing flows have been regularly performed twice a year (in autumn and spring). These flushing flows have provided opportunity to the design of such flows to increase their effectiveness, and macrophytes removal rates as high as 95% have been achieved in areas close to the dam (Batalla and Vericat, 2009). Despite the need to limit peak floods to avoid damage to riverine villages, flushing flows in the Lower Ebro are now a tested means to enhance the biological productivity of the physical habitat, to entrain and transport sediments to restore the dynamism of the river channel, to remove pollution loads and improve the water quality, to control salt intrusion and to supply sediments to the delta and the estuary.

Fig. 2 presents the standard hydrograph of the flushing flow implemented in the Lower Ebro since 2002 (for an extensive anal-

[1] Macrophytes are visible algae and other flora species that are rooted in shallow waters with vegetative parts emerging above the water surface. In lakes, macrophytes provide cover for fish and substrate for aquatic invertebrates, produce oxygen, and act as food for some fish and wildlife, therefore being a symptom of a good environmental status. However, in a river their proliferation occurs when water is stagnated and denotes a poor environmental status, having negative effects over the ecosystem and economic activities.

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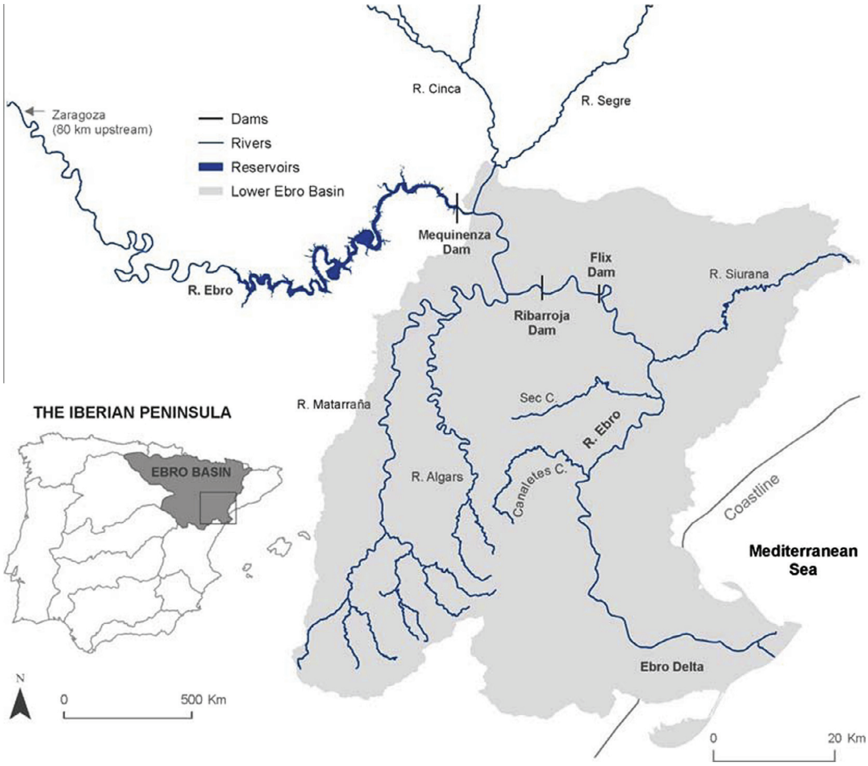

Fig. 1. Location of the River Ebro Basin in the Iberian Peninsula and detail of the Lower Ebro River. *Source*: Own elaboration from ERBA, 2012a.

Table 1
Characteristics of the Mequinenza–Ribarroja–Flix dam system.

Reservoir	Mequinenza	Ribarroja	Flix
Storage capacity (h m^3)	1530	218	5
Licensed flow (m^3/s)	760	940	400
Installed capacity (kW/h)	324	262.8	42.5
Height (m)	74	41	12.1
Efficiency	0.8	0.8	0.8
Input output ratio (m^3/kW h)	6.2	11.19	37.91

ysis of the flushing flow design and field monitoring, as well as a critical discussion on its effectiveness as a river restoration tool, see Batalla and Vericat, 2009).

3. Materials and methods

The opportunity cost of artificial flood flows in modified river reaches, where the flow regime is basically determined by the operation of hydropower facilities, can be defined as the reduction of the value of the energy produced resulting from the new environmental constraints. The assessment of this opportunity cost requires knowledge of the hydropower operator's profit maximising decision-making and how it would react to a change in the operating constraints imposed by the river basin authority. To solve this problem, we present a theoretical general model that allows the calculation of the opportunity cost of flushing flows based on the previously stated characteristics and we calibrate the general model to our particular case study in the Lower Ebro River[2].

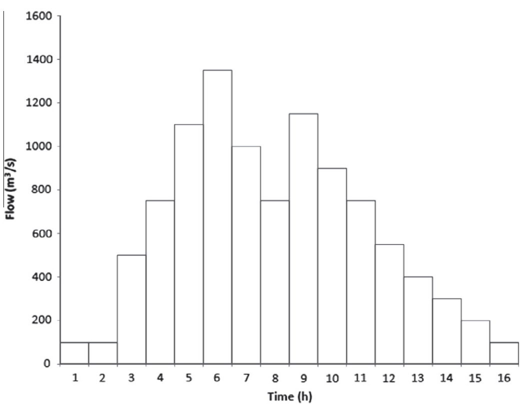

Fig. 2. Standard hydrograph of the flushing flow implemented in the Lower Ebro River since 2002. *Source*: Own elaboration.

[2] It is important to note that in the calibration stage we use econometric techniques. An analytical solution to the theoretical problem would demand accepting strong assumptions about the operator's behaviour (assuming either perfect hydrological foresight or accepting strong assumptions about the operator's risk attitude) that make preferable to deduce this solution from the decisions that the operator has taken in the past.

3.1. The basic opportunity cost evaluation model

From the hydropower operator's perspective, the dam and its associated power production facility are capital assets. At any given time, the operator decides on the flow of energy to be produced. This decision is based upon a number of variables, such as the technical characteristics of the plant, the current operating rules, the expected evolution of the amount of water stored in the reservoir and the current and the expected energy prices. From a private business perspective, these decisions aim at maximising the value of the expected flow of benefits along the entire life span of the dam. As the electricity produced cannot be stored for future selling, the hydropower operator has to make two kinds of decisions simultaneously. The first decision involves choosing how much water to release every day (x_t), and the second involves choosing how to distribute the electricity generated throughout the day (x_{tk}). Both decisions aim to maximise the flow of hydropower revenues. In what follows, we analyse each of these key production decisions:

Decision (1): the volumes of water released everyday can be represented by the following dynamic optimisation programme. For simplicity, we assume a zero discount rate:

$$\max_{x_t} \sum_{t=0}^{\infty} E[\Pi_t(x_t)] \tag{1}$$

$$z_{(t+1)} = z_t + y_t - x_t \tag{2}$$

$$\underline{z_t} \leqslant z_t \leqslant \overline{z_t} \tag{3}$$

$$\underline{x_t} \leqslant x_t \leqslant \overline{x_t} \tag{4}$$

where the decision variable x_t represents the flow of water used for power generation on day t; the function $\Pi_t(x_t)$ represents the daily financial revenue at the moment t, which (see below for details) is assumed to increase at a decreasing rate with the amount of water used to produce energy. The upper case E underlines the fact that companies' decisions are based on imperfect information concerning the future values of critical variables, such as the level of the reservoir and future energy prices (i.e., nature and market uncertainty imply that the objective function is in fact the expected value of the energy produced; thus, the model avoids the problem of most optimisation models that assume that companies have "perfect hydrological foresight", which leads to unrealistic results). The state variable z_t measures the amount of water stored in the reservoir on day t; its dynamics are represented by the transition function (2), where the state of the system on the following day depends, first, on its state the previous day, second, on the exogenous net inflow of water (y_t) obtained from the river basin net of the evaporation and the abstractions taken from the reservoir for other uses that are out of the control of the hydropower operator and, finally, on the decision made by the hydropower operator on day t, x_t.

Constraint (3) shows the boundaries of the state variable z_t on any day. The left term of this constraint shows the minimum level of water stored ($\underline{z_t} \leqslant z_t$). This lower bound is the value determined by the technical requirements of the infrastructure or by the institutional requirement to guarantee a minimum water availability for other present and future uses. Thus, the lower limit may vary in different seasons or months (depending, for example, on seasonal crops requirements). The right term of the constraint (3) shows the upper bound of the amount of water stored ($z_t \leqslant \overline{z_t}$), which also may depend on different factors, such as the reservoir's storage capacity or the flood limit to avoid the flooding of downstream riverine villages (which may also vary during the year according to flood risk perceptions).

Constraint (4) shows the boundaries of the daily decision variable, x_t. The lower bound ($\underline{x_t} \leqslant x_t$) may come either from a minimum environmental flow, from the requirement to deliver given amounts of water to other water uses downstream or, alternatively, from any water authority requirement to release a certain amount of water at the date t (for example, for an artificial flood). In a similar way, the relevant upper limit ($x_t \leqslant \overline{x_t}$) is the higher value among the quantity of water resulting from the hydropower generation plant maximum capacity. Provided that the plant is not always functioning at its full capacity, none of the above-mentioned constraints is binding, and the operator is able to distribute the energy produced among the different days of the year in order to maximise its revenue[3].

Decision (2) consists of choosing the hourly production of electricity in a particular day. This decision can be represented by the following daily revenue maximisation problem:

$$\max_{x_t} \Pi_t = \sum_{k=1} p_{tk} \alpha x_{tk} \tag{5}$$

$$\underline{x_k} \leqslant x_k \leqslant \overline{x_k} \tag{6}$$

$$\sum_{k=1}^{24} x_{tk} = x_t. \tag{7}$$

The objective function in this case, π_t, represents the daily financial revenue. This revenue depends on the following: (i) the decision variable (x_{tk}),the quantity of water used for power generation per hour (k), (ii) the corresponding prices, (p_{tk}), which are assumed to vary in a predictable way (t) depending on the season, the day of the week, weather conditions and other factors that are known in advance by the operator, and (iii) an input–output technical parameter (α) measuring the volume required to produce one unit of electricity. Under these conditions, the operator finds the optimal distribution of the energy produced during the day (producing at a maximum capacity at peak price and minimising the energy delivered to the market when electricity demand is at its lowest). The variable and fixed costs of producing hydroelectricity can be considered negligible; accordingly, variations in the revenue function reflect changes in financial returns. The decision variable (x_{tk}) is subject to the same upper and lower bounds as in the first problem, but the relevant time units are now hours instead of days (as in (6)).

Provided that there is detailed data on both the hourly market price of electricity and all of the relevant constraints on the decision variable x, obtaining a closed solution for decision problem (2) becomes straightforward. The solution of this problem for the range of all of the likely values of the daily decision x_t is the financial revenue function $\pi_t = F(x_t)$. This maximum daily revenue function is concave and non-decreasing and varies on different days during the year according to random and seasonal changes in electricity demand and supply.

Problems (1) and (2) are closely linked. On the one hand, the overall quantity of water delivered in the solution of problem (2) must equal the optimal decision of the first problem for the corresponding day (as in constraint (7)). On the other hand and most importantly, the optimal solution of problem (2) is nested in the definition of problem (1). In other words, the maximum revenue as a function of the decision variable (x_t) becomes the main argument, and its expected value in the future is the objective function of problem (1). Thus, when deciding how much water to use each

[3] In fact, the key role played by hydropower in the stabilization of the electricity supply system implies the presence of spare capacity ready to be used to turbine water at peak demand hours. In the last 10 years, hydroelectricity in Spain used less than 20% of its installed power production capital (Gómez, 2009).

day, the operator knows how this water can be delivered at any time to obtain the maximum revenue in the electricity market.

3.2. The model calibration

The maximum daily revenue function above is an important step in the calibration of daily production decisions as represented in problem (1). Nevertheless, finding the analytical solution to problem (1) is not an easy task given its dynamic nature, the wide time span that needs to be considered and the uncertainty associated with natural water inflows and energy markets. A theoretical solution requires assuming either perfect hydrological foresight or accepting strong assumptions about the operator's risk attitude. Instead of finding the analytical solution of problem (1), we have the option of deducing its solution from the decisions that the operator has taken in the past under a given set of conditions.

Therefore, we use detailed data on the decisions taken by the operator in the past (on different days, under different decision constraints, and in different states of the river system) to obtain econometrically the operator's underlying decision function of using water and producing energy. This function (problem (1)) and the maximum daily revenue function (problem (2)) provide the representation of the optimal behaviour of the operator in the baseline scenario. These two functions and the information set of observed decisions and constraints are all that we need to represent the operator's behaviour and assess the opportunity cost of imposing the delivery of a flushing flow.

The information used in this paper comes first from the daily data on the level of water stored in the three reservoirs and their hourly outflow of water provided by the Hydrological Information Automatic System (SAIH) of the Ebro River Basin Authority (ERBA, 2012a). We use data from September 1997 to October 2008. This 11-year period encompasses several hydrological cycles during which regulations over water use have been relatively stable, as defined in the River Basin Management Plan (ERBA, 2012b). Secondly, the River Basin Authority has also provided an entire set of data on the relevant constraints with which the operator must comply. These data include the following: the minimum flows, set at 100 $m^3\ s^{-1}$; the amount of water that was required to be supplied by the reservoir system for other uses different from power generation in any given month; and the monthly changing minimum level of water stored in the MRFDC determined by the water authorities to guarantee water supply at any time. Finally, the hourly price of electricity was obtained from the Spanish Electricity Market Operator (SEMO, 2013), and the quantity of electricity produced by the hydropower operator at any moment was deduced from the outflow of water and the technical characteristics of the power plants in each reservoir (we assume a standard 0.8 energy conversion efficiency). In this way, we have observations for all the parameters and for all the state and decision variables implied in the optimisation problems (1) and (2) for a total sequence of 4017 days. This sample provides both the data required for calibrating the base model and the scenario to assess the opportunity cost of the flushing flow programme.

The *first stage* in calibrating the model deals with optimisation problem (2). The daily financial returns are a maximum argument function of the following: the amount of water used for power generation, the set of hourly prices of the day, the minimum flow set by water authorities and the maximum production capacity of the plant. Fig. 3 shows the daily financial return function obtained from using hourly prices and the production capacity and the minimum flows for three selected months: (i) December, when water demand and the average price are that their highest, (ii) March, when prices are the lowest, and (iii) January, when the price is close to the yearly average. As can be observed, the financial return function increases at a decreasing rate with the volume of water.

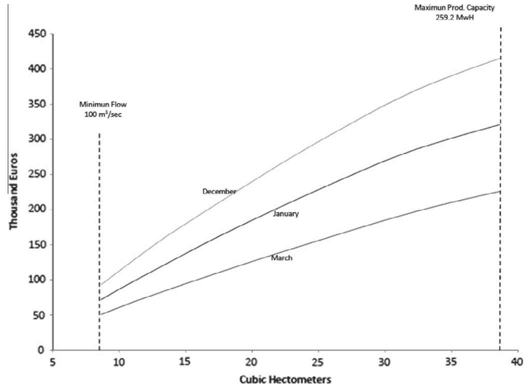

Fig. 3. The optimal daily revenue function in the upstream Mequinenza Power Plant. *Source*: Own elaboration.

Once the minimum flow is satisfied, the decreasing marginal productivity of the water input is caused by the fact that at lower production levels, the energy is produced at peak price time; any increase in water use implies selling the energy at a decreasing price. Daily income is also bounded by the maximum capacity of the plant.

Once the optimal financial returns are determined, this information is introduced in the intertemporal decision problem (1) to obtain the optimal decision profile of how much water to use any day, considering the transition equation (2) and the technical and policy constraints of the baseline scenario. The ability of the operator to obtain rents from market price variations is one of the key elements that are affected by the requirement to adjust water delivery to a pre-designed flushing flow scenario.

The *second stage* of model calibration deals with optimisation problem (1), which is associated with the decision on the daily outflow of water. Obtaining an explicit functional form of the optimal daily decision profile x_t is not feasible given the number of parameters involved and the stochastic nature of the problem. Nevertheless, the number and the details of the available data in the sample allow for an empirical approximation of this optimal value function with econometric techniques; this circumstance allows revealing the functional form that better explains the observed behaviour of the operator. We thus expect the decision variable (x_t) to be an increasing function of the amount of water stored (represented by the state variable z_t) and the water inflow received from the basin on the previous days, y_t. As this relationship is not linear, we use a maximum likelihood estimation method to obtain the better fitting function among the Box Cox power transformation family of functions. In addition, as restrictions over the minimum level of the stored water and the other uses of water that are different from electricity production vary from month to month, we also used dummy variables for every month of the year. The empirical model is then as follows:

$$x_t(\mu) = \alpha_1 z_t(\lambda) + \alpha_2 y_t(\eta) + \beta_i, \tag{8}$$

where λ, μ and η are the Box Cox transformation parameters:

$$x(\mu) = \frac{x^\mu - 1}{\mu} \cdot z(\lambda) = \frac{z^\lambda - 1}{\lambda} \cdot y(\eta) = \frac{y^\eta - 1}{\eta} \tag{9}$$

and the coefficients $\beta_i (i = 1, \ldots, 12)$ represent the fixed effect parameters for any month that is included in the model as dummy variables. The variable y_T measures the overall net inflow from the

Table 2
Box Cox estimation of the daily outflow of water. *Source*: Own elaboration.

Variable	Coefficient	Standard error	Significance (%)
Water stored (h m^3)[a]	0.0009339	0.0368078	99
Lag water stored (h m^3)	0.1480627	0.00718118	99
Water inflow (h m^3/day)	0.48437	8.13874E−05	99
October	−12.4484	1.34145119	99
November	−11.278	1.34805527	99
December	−10.647	1.3814159	99
January	−8.71131	1.39861698	99
February	−9.53085	1.40190417	99
March	−8.7341	1.41103021	99
April	−10.3591	1.44114208	99
May	−10.9435	1.47712765	99
June	−13.4476	1.48860571	99
July	−12.3389	1.44571131	99
August	−12.4329	1.37090634	99
September	−12.8491	1.32810979	99
λ	0.383011	0.00718118	99
Wald test	34.07		
Elasticity of water stored	1.14593		
Elasticity of water inflow	0.79495		
Elasticity of lag water inflow	0.35503		

[a] Variables transformed by λ.

upstream river basin and helps to include variations explained by dry or wet years.

This function of the private decision on how much water to deliver on any day, along with the maximum revenue function determining how to distribute this water during the day to produce energy, allows the calibration of the model for the complete sequence of all of the days in the sample. Table 2 shows the econometric results. Transformation parameters μ and η were not found to be significantly different from 1; therefore, the associated variables x_t and z_t enter linearly in the equation. The maximum likelihood value of the nonlinear transformation parameter (λ) was determined at 0.35. All of the remaining coefficients are significant at a 1% level. Apart from maximum likelihood criteria, the final equation fulfils Wald's and Lagrange's multiplier tests for the optimisation of the econometric estimation. The size and detail of the sample seem to be important factors behind the robust and efficient econometric estimation of the daily decision variable.

This baseline scenario and the associated optimisation functions are the basis by which to assess the impact of flushing flows over the quantity and value of the energy produced.

4. Results

Flushing flows are implemented through the imposition of particular constraints over the operating rules of the hydropower plant. These constraints imply a deviation from the profit maximising decision profile (baseline scenario) with a negative impact on expected financial profits. The revenue variation, or the opportunity cost, is moreover the net result of two different effects of opposite sign. The first effect is the immediate revenue increase, as controlled floods require the delivery of an amount of water that exceeds the quantity that the operator would have chosen otherwise. The second effect is the decreased revenue resulting from the reduction in the stock of water available after the flood[4] during the days or weeks required for the reservoir to come back to its baseline level. Once this convergence is complete, not only will the amount of water stored be back to normal but the operator's decisions and revenues will also be the same as in the baseline scenario. The absorption period, or the time during which water stocks, flows and profits diverge from the baseline, is a measure of the time required by the system to absorb the shock produced by the flood[5].

The cost of the flushing flow can be reduced by a careful selection of the right moment at which to start delivering the water for the subsequent hours. Although the operator cannot decide upon the day and the quantity of water to deliver during the artificial flood, it can choose the right hour at which to start the flood. This decision allows minimising foregone revenues, as expected energy prices vary in a predictable way during the day. Fig. 4 shows the market value of the energy obtained during the flood for the autumn and spring seasons according to the flushing flow hydrograph shown in Fig. 2.

The correct selection of the time to start delivering the water might explain differences as high as 40% of the maximum revenue, or an opportunity cost of as much as EUR 160,000 per flood. In what follows, we assume that the delivery of water always starts at a time that maximises the value of the energy produced during the flushing flow (thus minimising the opportunity cost of the flushing flow).

Provided that the artificial flood is feasible (which occurs when water level is above a minimum critical level) and its starting point has been chosen to minimise its impact over the value of the electricity produced, we are now ready to analyse the opportunity cost of flushing flows. Fig. 5 presents the overall opportunity cost for the days in the sample when the flood is feasible in autumn (Fig. 5a) and spring (Fig. 5b). The revenue variation is measured on the left axis. The figure is complemented with data about the amount of water stored in the upstream reservoir on the day of the flood, which is measured on the right axis.

As expected, the opportunity cost of flushing flows changes with the condition of the system. The profit maximising opportunity cost varies from EUR 33,000 (revenue variation: −33,000) to EUR 76,000 (−76,000) for the spring and autumn floods, respectively, for a total opportunity cost of 109,000 EUR per year (−109,000). The standard deviation of the opportunity cost equals 55,000 for the spring flood and 110,000 for the autumn flood, denoting a high variability that is largely the result of the irregular water flows observed in the case study area. In the same way, the absorption time varies from a few days to several months with an average value of 82 days and a standard deviation of 50[6].

5. Discussion and conclusions

Flushing flows have been shown to be effective means to achieve successful river restoration (Hueftle and Stevens, 2002;

[4] Under extreme events, the implementation of flushing flows may lead to additional opportunity costs. For example, when the amount of water stored in the reservoirs is below or at its lowest or minimum acceptable level, flushing flows would imply a reduction of the water supplied for crops or any other uses. In any case, despite being technically feasible, the River Basin Authority clearly establishes a series of priorities under extreme events that rule out the possibility of implementing flushing flows (ERBA, 2007).

[5] The flushing flows alter the decision making process of the hydropower operator, moving away the observed stocks and released water flows from the optimal path (baseline). As a response to the lower water stock in the dam resulting from flushing flows, the hydropower operator will release less water than he would otherwise do in the baseline scenario without flushing flows. This will happen until the amount of water stored in this alternative scenario finally converges to the amount of water stored in the baseline scenario. This time span is known as the absorption period.

[6] Operator's decisions in our model are based on expectations over the water inflow that the reservoir might receive in the future. These expectations might or might not be fulfilled, and the consequence of this circumstance is that the opportunity cost may actually differ from its expected value (depending on rainfall on the days following the flood) and can even be negative. Given the timing of the different effects and, particularly, the fact that the increase in revenue occurs at the start of the flood, while the cost is different along the absorption time, the succession of wet days can shorten the absorption time, and when the reservoir recovers rapidly enough, it can even avoid a negative opportunity cost. This outcome is observed on the days when the revenue variation of the flood is positive (see Fig. 5).

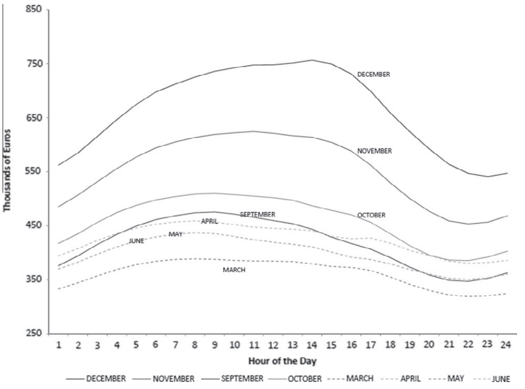

Fig. 4. The optimal timing of the flushing flow in the Lower Ebro River. *Source*: Own elaboration.

Vinson, 2001; Kondolf and Wilcock, 1996). While the benefits and technical effectiveness of this alternative are widely known, the tradeoffs in terms of the economic uses of water are often considered too high and prevent the periodical release of artificial floods. In this paper, we present a planning-level methodology for the assessment of such opportunity costs in heavily modified downstream areas where flushing flows affect the operational rules of hydropower facilities. We show how the model can be calibrated with a combination of a deterministic maximum revenue function for the hourly delivery of water and an econometrically obtained decision function for the daily amount of water delivered. The model enables us to analyse the impact of imposing a new operation rule on the hydropower operator's optimal decisions. This rule requires the release of water during certain periods of time in accordance with an artificial flood regime purposely designed to restore the basic functions of a river ecosystem. As the technical design, feasibility and opportunity cost of flushing flows heavily depend on the intrinsic conditions of river ecosystems, we used the detailed time information about the stocks and flows of water in the Lower Ebro River to calibrate and simulate the model for all of the days in spring and autumn in the sample when an artificial flood is feasible.

Implementing flushing flows on a regular basis will result in a reduction in the asset value of affected hydropower facilities, as they will have to operate under more stringent institutional rules. The case study shows that hydropower facilities in the Lower Ebro can provide the artificial flows required for the restoration of the river channel at a cost that is equivalent to a small fraction of the energy delivered to the market and the overall annual revenue. The expected opportunity cost of two floods per year (EUR 109,000) is equivalent to 0.17% of the average yearly revenue and is only a fraction of the average daily revenue (which amounts to EUR 250,000 in the sample days).

The cost of guaranteeing the periodical release of flushing floods by changing the operation rules of hydropower facilities also seems to be lower than any other alternative of obtaining water from other sources (such as saving water in agriculture and domestic consumption or water recycling and desalination) to have additional stored water available for this purpose in the reservoirs. Each artificial flood requires the delivery of approximately 36 million cubic metres over sixteen hours; considering the opportunity cost estimated at EUR 76,000 and EUR 33,000 for the autumn and the spring floods, respectively, we can conclude that the cost per cubic metre delivered is lower than EUR 0.002 for the autumn flood and less than half of that quantity for the spring flood. Experience shows that there are few alternatives to obtaining such a large amount of water at a lower cost from other economic uses.

Provided that flushing flows are implemented with sound economic criteria, their opportunity cost is small when compared to people's Willingness To Pay (WTP) to secure the benefits of river restoration programmes. Original estimations in areas that resemble our policy context show that WTP ranges from EUR 5.3 to EUR 63.6 per person per year (Loomis et al., 2000; Meyerhoff and Dehnhardt, 2007; Berrens et al., 1998; Brown and Duffield, 1995; Colby, 1993; González-Cabán and Loomis, 1997). Depending on the size of the population benefited by the programme, the opportunity cost of flushing flows can range from EUR 0.55 (if we consider the 200,000 people living in the Lower Ebro River) to EUR 0.04 per person per year (if we consider the 2.8 million people living in the entire Ebro Basin) (ERBA, 2012b).

However, these values should be taken with caution. The WTP for the benefits associated with river restoration programmes may be actually lower as a result of the distance decay problems typically associated with environmental quality valuation (Hanley et al., 2003; Bateman et al., 2006). Also, the opportunity cost of flushing flows of 109,000 EUR per year should not be regarded strictly as a lower bound; rather, it is a reference value sensitive to uncertainty. Actually, the uneven behaviour of flows and stocks of water in Mediterranean rivers (ERBA, 2012a) and the volatility of energy prices (SEMO, 2013) make operator's revenue highly variable. Assuming that hydropower operators are risk averse, they would be willing to accept a lower compensation for the losses derived from a flushing flow, as long as this value is secure (*certainty equivalent*). The difference between this compensation and the opportunity cost of flushing flows is a function of the operator's risk aversion coefficient, which varies for every area and type of agent (e.g., risk aversion is higher in drought prone areas such as the Guadalquivir River Basin than in more resilient basins, see Gutiérrez-Martín and Gómez, 2011). Although revealing the risk attitude of hydropower operators is beyond the scope of this paper, these considerations need to be addressed in future research and bargaining processes.

In spite of this, our results show that the opportunity cost of flushing flows is expected to be between 9.74 and over 1633 times lower than the benefits associated with the river restoration programmes, as measured by individual's WTP. These figures suggest that the real policy challenge consists in finding the institutional agreement to implement the flushing flood programme and agreeing on the potential compensations[7] to overcome the incentive problem. The considerable mismatch between the opportunity cost and the societal benefits provides sufficient room for private operators and public authorities to conduct successful bargaining and thus agree on the voluntarily compliance of a soundly designed programme of flood releases to restore the critical functions of the water ecosystem. The cooperation between power generation companies and water authorities is also a positive signal, showing that flushing flows for river restoration purposes can be compatible with private corporate interests. These efforts are now considered to be the pioneering phase of a comprehensive restoration programme of the whole river's ecosystem and a key piece of the River Basin

[7] The MRFDC was built by the hydropower operator in exchange of a long term government concession to exploit the dam complex. The contract did not include the possibility of implementing flushing flows (i.e., larger temporary outflows), nor a reduction in the water inflows. In the past, the modification of this contract (e.g., through increased water rights upstream that reduced water availability for hydropower generation) has been solved with compensations to the operator, sometimes in the form of larger concession periods (e.g., GRC, 1997). Accordingly, in the current context making the hydropower operator pay for the flushing flows cost is unlikely and would require a modification of the legal framework.

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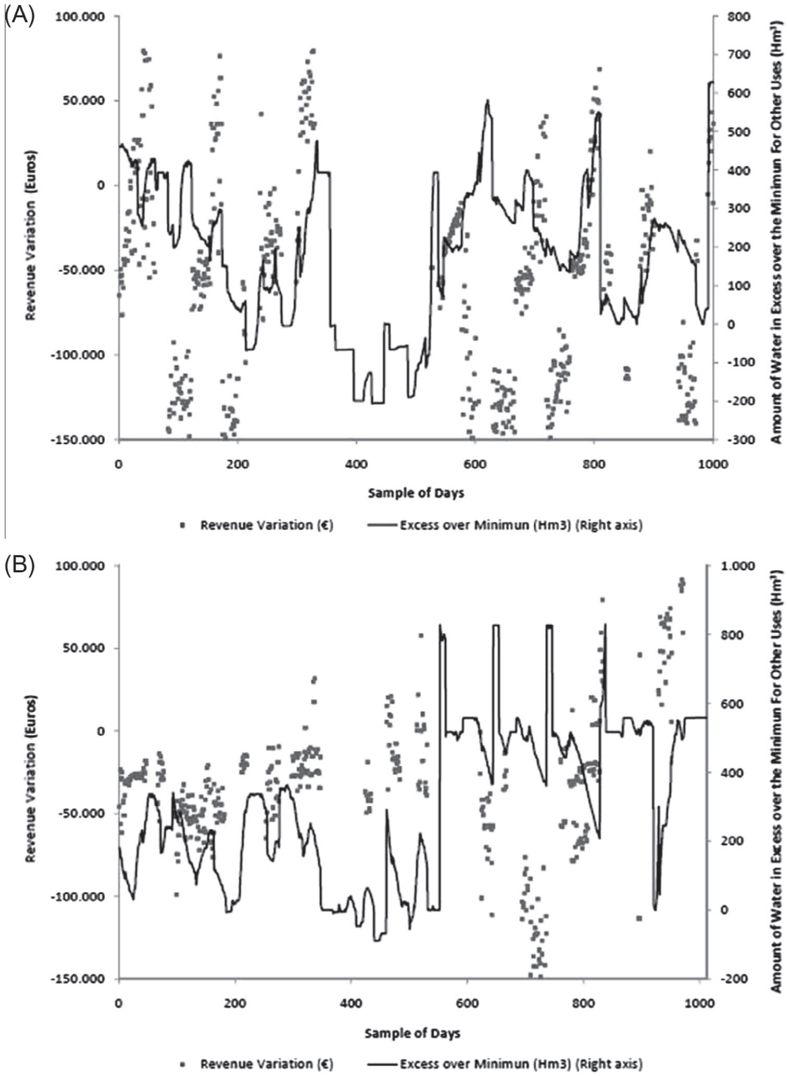

Fig. 5. The opportunity cost of a flushing flood in the Lower Ebro. (A) Autumn. (B) Spring. The sample of days includes the sequence of autumn (A) and spring days (B) in the total sequence of 4017 days (ERBA, 2012a). *Source*: Own elaboration.

Management Plan that is being elaborated for the implementation of the WFD.

Also, this research on the opportunity costs of flushing flows may offer useful insights for basins that resemble our case study area. There is still little research on the costs of reallocating water from economic uses to the environment, with the exception of some studies on the tradeoff between agriculture and environmental flows (Sisto, 2009; Troung, 2012; Pang et al., 2013). However, as shown above, the implementation of flushing flows in heavily engineered rivers like the Lower Ebro River may be more cost-effective if the necessary water is taken from alternative uses with a lower opportunity than agriculture (i.e., hydropower), provided that other uses are not affected. Moreover, the large amounts of water required and the short time span during which flushing flows are released may make hydropower the only feasible alternative. This paper aims to provide a standard method to estimate the tradeoff between flushing flows and hydropower generation. This methodology may be transferred to other heavily engineered rivers in which hydropower facilities can be used to reproduce the functions previously performed by the natural system and thus to achieve a better ecological status. This is the case of many rivers in semi-arid areas, which tend to be more heavily impounded and thus their hydrology more strongly affected than rivers in humid climates because demand for water is greater and runoff is out-of-phase with demand. For example, in the Sacramento and San Joaquin Rivers of California (US) the impounded runoff index (ratio of reservoir capacity divided by mean annual runoff) is 0.8 and 1.2, respectively, and the flood peaks have declined on average

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53% and 81%, respectively. Therefore, flushing flows have the potential to achieve a better environmental status (Kondolf and Batalla, 2005). In these rivers runoff is lower than in the Lower Ebro River, pressures are more intense and the hydrograph is flatter (the decline of the flood peaks is estimated at 30% in the Ebro River) (ERBA, 2012a; Kondolf and Batalla, 2005), all this suggesting larger opportunity costs and absorption periods for flushing flows than in our case study area (but also potentially larger environmental benefits), though all this should be confirmed with on-site estimations. Similar results could be expected with the flushing flows proposed by Wu and Chou (2004) in the Trinity River in northern California. The estimation of the opportunity costs of flushing flows is also of relevance in the lower stretches of the Colorado River (US-Mexico border), where the recently approved Minute 319 created a pilot programme that required water users in the U.S. and Mexico to provide a one-time high-volume flushing flow (or *pulse* flow) of 129.5 million cubic metres (IBWC, 2012). However, since water scarcity is much more acute in this area (the delta of the Colorado River has run dry during most of the last half century) (Glenn et al., 2008; Wheeler et al., 2007; ERBA, 2012a), opportunity costs are likely to affect other uses apart from hydropower generation and therefore a more extensive assessment framework involving other economic activities would be required in this case. Flushing flows have also been implemented to prevent algal blooms downstream the Opuha Dam in New Zealand (Lessard et al., 2013), though with limited results as a consequence of the inability of the dam to generate floods similar to pre-dam levels. This area resembles our case study, with hydropower being the most affected economic activity. In this and similar cases, the estimation of the opportunity costs is of especial relevance in order to justify (or not) the implementation of flushing flows from a cost-benefit perspective.

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5.2 Drought insurance for irrigated agriculture

Droughts are a relevant temporary decrease of the average water availability and are considered natural phenomena. There are two types of droughts: meteorological and hydrological. Meteorological droughts refer to a precipitation deficit over a period of time, while hydrological droughts refer to unusually low water levels in reservoirs, river flows, streams, lakes and/or aquifers. There is a time lag between the lack of precipitation and decreased water levels, which makes that the end of a hydrological drought might also be lagging behind the end of the corresponding meteorological drought, since large quantities of precipitation are required to restore water bodies back to normal conditions (EC, 2008).

Droughts may have significant effects over the economy. An abrupt fall in the amount of water available implies that not all the water uses (agricultural, industrial, environmental and households demand) can be fully satisfied during a period of time, thus generating relevant welfare losses. Traditionally, policy makers have reacted to drought episodes through a crisis-management approach, rather than through the development of comprehensive, long term and planned drought policies. Although this is changing (e.g., through DMPs –see Chapter 4), the uncertainty associated with this emergency response has promoted spontaneous and individual actions to enhance the preparedness against droughts. These actions often involve significant tradeoffs with negative environmental impacts. This is especially visible in the case of agriculture, the world’s largest water consumer (Ward and Pulido-Velazquez, 2008).

Farmers are risk averse individuals that are willing to reduce their expected income as long as this income becomes more stable (Gutierrez-Martin and Gomez, 2011; Lien and Hardaker, 2001). In arid and semi-arid areas, the most important threat to agricultural income stability is droughts (OECD, 2010). Accordingly, farmers are willing to pay in advance relatively large amounts of money (as compared to their income) to soften the impact of droughts. In rainfed agriculture, this can be done through drought insurance[20] (Bielza et al., 2008b). However, drought insurance for irrigated agriculture does not exist in the EU (Bielza et al., 2008a, 2008b). As a result, especially in drought prone and highly profitable agricultural areas, farmers may incur in informal abstractions from loosely controlled groundwater bodies to

[20] Currently only a few countries offer drought insurance for rainfed agriculture in the EU, namely, Austria, France, Italy and Spain.

stabilize their income (Pérez-Blanco and Gómez, 2013). This may be one of the main factors driving overexploitation and aquifer depletion in many Mediterranean basins (EC, 2008; EEA, 2009; WWF, 2006). Our main hypothesis is that an insurance system provided by financial markets may allow transferring the burden of drought risk from nature to the financial sector, thus removing this negative tradeoff. Stabilizing farmers' incomes then becomes a way to reduce incentives to informally abstract water from overexploited aquifers[21].

The implementation of a drought insurance system for irrigated agriculture poses many challenges. However, Spain has exceptional enabling conditions as proved by the success of the agricultural insurance sector in covering a wide range of natural hazards. In fact, Spain has nowadays the most developed agricultural insurance sector in the entire EU (Bielza et al., 2008a, 2008b). Moreover, most of the challenges faced by drought insurance for irrigated agriculture resemble those faced and overcome by drought insurance for rainfed agriculture, and therefore could be easily dismantled (Pérez-Blanco and Gómez, 2013). Accordingly, it seems that the failure to successfully implement drought insurance for irrigated agriculture has to be more with a lack of institutional development and the fears surrounding its financial implementability.

The more practical way to identify the room available for the development of this economic instrument consists in estimating the maximum welfare surplus at stake. These gains are the difference between the maximum amount farmers are willing to pay for the insurance and the minimum costs at which this product may be provided by the financial market. This surplus is usually positive because firms can aggregate individual risks and are risk neutral while individual farmers are risk averse.

The total costs faced by an insurance company comprise the expected indemnity, transaction costs, asymmetric information and systemic risk costs. Transaction costs are in principle negligible since drought insurance for irrigated agriculture would be offered as a part of a comprehensive insurance package covering other agricultural hazards, known as the *combined insurance scheme*. Therefore, the increase in transaction costs by drought insurance for irrigated agriculture would be only marginal. However, in water scarce and drought prone areas the expected indemnity (the expected value of the yield losses effectively compensated by the

[21] Precisely because of this, it is of paramount importance that drought insurance for irrigated agriculture focuses at least on the farmers whose plots have access to overexploited groundwater bodies.

insurer[22], measured in constant prices) can be high, thus increasing the total costs and reducing the room to cover asymmetric information and systemic risk costs. In addition, drought insurance is especially sensible to asymmetric information (i.e., moral hazard and adverse selection) and systemic risk problems, which may result in the total costs overcoming the farmers' willingness to pay.

In the following papers we estimate the fair risk premium of drought insurance for irrigated agriculture for a varied range of ligneous crops. The fair risk premium is a key variable in any insurance system and equals the quotient of the expected indemnity to the production value in a hydrological year without drought. In other words, it represents the direct costs of the drought in the case of irrigated agriculture, or alternatively the long term costs of providing this type of insurance in a world without information constraints and transaction costs. This value, combined with the farmers' willingness to pay (Gutierrez-Martin and Gomez, 2011), can be used to estimate the room that insurance companies have to accommodate transaction costs and especially the costs of asymmetric information and systemic risk.

This chapter presents two papers, both authored by the doctoral candidate and the Prof. Carlos Mario Gómez Gómez. The first one is entitled *Designing optimum insurance schemes to reduce water overexploitation during drought events: a case study of La Campiña, Guadalquivir River Basin, Spain* and was published in the Journal of Environmental Economics and Policy in 2013. This paper was presented in the conference *The Governance of Sustainability* in Cambridge, UK (11-12 April 2012); and in the *International Water Resource Economics Consortium (IWREC) 10th Annual Meeting* in Stockholm, Sweden (26-31 August 2012).

The second one is entitled *Insuring water: A practical risk management option in water scarce and drought prone regions?* and has been accepted for publication in Water Policy in 2013. This paper was presented in the *123rd EAAE Seminar. Price volatility and farm income stabilization* in Dublin, Ireland (23-24 February 2012); in the *86th Annual Conference of the Agricultural Economics Society* in Warwick, UK (16-18 April 2012); in the *Vth AERNA Conference* in Faro, Portugal (31 May-2 June 2012); in the *International Society of Ecological Economics 2012 Conference – Ecological Economics and Rio+20* in Rio de Janeiro, Brazil (16-19 June 2012); and in the *Belpasso Summer School: Frontiers in Economics of Natural Hazards and*

[22] Insurance systems compensate only a fraction of the yield losses. This avoids a full loss recovery and reduces the incidence of moral hazard.

Disaster Risk Reduction - Financing Disaster Risk Reduction and Climate Adaptation in Belpasso, Italy (1-7 September 2013).

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Designing optimum insurance schemes to reduce water overexploitation during drought events: a case study of La Campiña, Guadalquivir River Basin, Spain

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In several arid and semi-arid Mediterranean basins, water deficits in irrigated agriculture during drought events are relieved by illegal abstractions from aquifers. Illegal abstractions are largely tolerated by the authorities and are regarded by farmers as a reliable and inexpensive form of insurance against drought. This framework of illegal abstractions is responsible for the structural water deficit that is characteristic of many Southern European regions. The situation is changing with the implementation of River Basin Management Plans and Drought Management Plans, which demand improvement in the quantitative and qualitative status of water bodies, improved surveillance of groundwater resources and more rigorous sanctions for illegal groundwater abstractions. However, these plans raise distribution and equity issues and may not be sufficient to stop illegal abstractions in certain areas. Provided that the new framework is properly enforced, private drought insurance has the potential to stabilise income levels and reduce the incentives for overexploitation during drought events. This paper develops a methodology to estimate the basic risk premium and the potential water savings of private drought insurance. This methodology is based on concatenated stochastic models (rainfall-stock), a decision model and agronomic production functions, and is illustrated through the application of the model in the La Campiña agricultural district in the Guadalquivir River Basin, Spain.

Keywords: drought insurance; stochastic models; groundwater; agriculture; drought management plan

1. Introduction

Fresh water is a finite and vulnerable resource that is essential for sustaining life, development and the environment and should be managed accordingly (ICWE 1992). However, the prevailing paradigm has been that water policies must play an instrumental role in providing a package of services, thus making water demand exogenously defined outside of the field of water management policy (Saleth and Dinar, 1999). In the case of Southern Europe, this paradigm has led to a significant

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expansion of irrigation, which has increased the pressure on water resources. This increasing demand contributes to the arid and semi-arid climates that are characteristic of many of these regions, which significantly constrains water availability. Consequently, this area is now more vulnerable to droughts. In an attempt to limit the impact of droughts on their activity, farmers have reacted by increasing illegal abstractions from uncontrolled, dependable aquifers. The reluctance of the water authorities to penalise this behaviour has made illegal abstractions *inexpensive*, and these abstractions have become the *true* insurance against drought.

The Water Framework Directive (WFD) was implemented as a reaction against the poor qualitative and quantitative status of the water bodies, particularly aquifers. Following the WFD, all members of the EU must have approved River Basin Management Plans (RBMPs)[1] by 2009, which include the achievement of a good quantitative status in every relevant water body as a priority. The European authorities also recommended the development of Drought Management Plans (DMPs) in drought-sensitive basins. Drought Management Plans are intended to avoid water overexploitation during drought events through a set of objective drought indicators and abstraction rules[2] (EC 2008). Both RBMPs and DMPs indicate a clear commitment to stop illegal abstractions during drought events through an improved surveillance mechanism and a more rigorous application of sanctions (GRBA 2007, 2010). Consequently, the likelihood of detecting and penalising offenders increases, which may provide a sufficient incentive for farmers to reduce illegal abstractions, or even to stop them, in areas where the income gap between irrigated and rain-fed agriculture is small (Mendelsohn and Saher 2011). However, this incentive may prove to be insufficient in agricultural districts where the income gap is large and droughts result in significant monetary losses (WWF/Adena 2006, Llamas 2007).

Drought insurance guarantees a regular income and thus implements incentives to reduce illegal abstractions, even in high-income irrigated areas. However, drought insurance for irrigated agriculture does not exist in Europe[3] (Bielza *et al.* 2008a, 2008b) for a variety of reasons. The systemic nature of droughts, moral hazard and adverse selection are frequently cited problems that raise the price of the product (Miranda 1991) or even make a certain degree of public support necessary (Rejda 2008), as happens with drought insurance in rain-fed agriculture (Bielza *et al.* 2008a, 2008b). Such problems are also common to other sources of risk where insurance markets are nonetheless working (World Bank 2005, Bryla and Syrpka 2007, Dick 2007, Breustedt *et al.* 2008). In the case of irrigated agriculture, the reason that drought insurance does not exist is twofold: (i) there is a high uncertainty that stems from *institutional decisions about water availability*, and (ii) there is a high *cost of insurance* compared to the alternative of illegal abstractions.

Prior to the implementation of DMPs, irrigation restrictions during a drought event were subject to discretionary assessments made by institutions. The outcome of the assessment was unpredictable and increased the indemnity variability above levels that would be bearable by commercial insurance (Pérez Blanco *et al.* 2011). Therefore, any compensation for farmers would come from public institutions through expensive emergency funds (Meuwissen *et al.* 2003). Moreover, emergency funds did not always guarantee a full refund of the losses. The high uncertainty made farmers perceive illegal abstractions from dependable aquifers to be a *more reliable* form of insurance with the potential to guarantee up to 100% of the yield. In addition, poor surveillance mechanisms, the tolerance of offenders and the small chance of being punished by the authorities made illegal abstractions *less expensive* than traditional insurance. For instance, the solution to persistent overexploitation has very often consisted of granting new water use rights to the offenders, thus generating additional incentives for further water overuse (Gómez Gómez and Pérez Blanco 2012). Under these conditions, the development of an insurance market for irrigated agriculture has not been possible.

If properly enforced, RBMPs and DMPs will increase the cost of illegal abstractions and largely remove institutional arbitrariness from decisions on water availability, thus making drought insurance relatively less expensive and more reliable. All this favours the development of an insurance system that is partially supported by private capital and more efficient and effective than emergency compensation, where *the informal, spontaneous and individual insurance system of illegal abstractions can be replaced by a more coherent, formal collective risk-sharing scheme*. This paper develops a methodology to explore the financial viability of drought insurance markets in irrigated agriculture and the potential water savings that can be obtained under the new framework characterised by DMPs and RBMPs. The potential of this methodology is illustrated by its use with the irrigated ligneous crops in the La Campiña agricultural district in the Guadalquivir River Basin (GRB), Spain. The results indicate that the basic risk premiums would be reasonable and the expected environmental outcomes significant.

2. Case study background: La Campiña, GRB, Spain

Spain has the most developed agricultural insurance system in Europe, in which all companies operate within a pool and assume the risk in a co-insurance regime (Bielza *et al.* 2008a). Spain has also pioneered the introduction of DMPs in the EU, and every relevant basin has approved its own DMP, including the GRB (EC 2008). Although the approval of RBMPs in Spain has been delayed, the GRB's RBMP is in its last stage, and there is a preliminary report available (GRBA 2010). The implementation of DMPs and RBMPs is part of a national strategy organised by the River Basin Authorities to provide a collective response to the increased frequency and intensity of droughts, especially in the south (Pérez Blanco *et al.* 2010).

Water demand in the GRB comes mainly from agriculture. Irrigation demands an average of 3485 hm^3 every year and represents 86.8% of the 4016 hm^3 annual water demand. However, renewable resources amount to only 3287 hm^3/year, and this value is reduced to 3028 hm^3/year if we consider the minimum environmental requirements (GRBA 2007, 2010). As a result, water overexploitation in an average hydrological year amounts to 987.7 hm^3, according to official records, with a water abstraction to renewable resources ratio of 1.22, which is considered severe overexploitation. Other sources estimate this ratio to be 1.64 (EEA 2009).[4]

La Campiña, located in the GRB in the south of Spain, comprises one of the most important agricultural districts in Spain. La Campiña Agricultural District has 48,764 ha of irrigated land, of which 39.7% is ligneous crops (GRBA 2010). The most relevant ligneous crops in the area are *Prunus armeniaca* (315 ha), *Cerasus* (4685 ha), *Citrus* × *sinensis* (408 ha), *Olea europaea* (9087 ha) and *Malus domestica* (997 ha). Greater than 99% of the water demand in the area is from irrigation. During an average hydrological year, 78.8% of water for irrigation comes from aquifers, with the remaining resources coming from reservoirs. There are three aquifers in La Campiña, all of them are detritic. The two main aquifers, whose renewable resources amount to 92.8 hm^3/year, have an abstraction-to-renewable resources ratio of greater than 100% in an average hydrological year. The third aquifer (32 hm^3/year) has an exploitation rate of slightly greater than 70% (GRBA 2010). In addition, droughts in the area are common and expected yields are high (Pérez Blanco *et al.* 2011), which makes the aquifers of La Campiña extremely vulnerable to drought events.

3. Methodology

The viability of an insurance market depends on the experimental design of feasible scenarios, the financial losses associated with these scenarios and the corresponding probabilities from which the risk premium is estimated (Skees and Barnett 1999).[5] The basic risk premium is the key element in the design of commercial insurance and is calculated as the ratio between the expected indemnity (a function of the expected losses) and the expected production value in a reference year (in this case, a normal or average hydrological year). The basic risk premium should not be confused with the premium applied in the insurance market, which is actually larger. This discrepancy between the basic risk premium and the insurance market premium occurs because of three factors inherent to agricultural insurance markets: (i) first, insurance markets are plagued with issues of adverse selection and moral hazard that may significantly increase costs for the insurer (Miranda 1991);[6] (ii) second, insured agents are risk averse (Lien and Hardaker 2001, Kim and Chavas 2003), and their willingness to pay to transfer a portion of the risk they bear to an insurer is greater than the expected drought losses;[7] (iii) finally, the implementation of an insurance system requires that the insurer constitutes a financial fund in which stochastic indemnities are compensated by the money paid by the insured, and this fund has intrinsic operating costs that are assumed by the insurer and must be recovered. In addition, drought is a systemic risk and is likely to generate catastrophic events with disproportionate costs for which commercial insurance may not be prepared; thus, a certain degree of public support is necessary (Rejda 2008, Bielza 2008a, 2008b).

This methodology calculates the basic risk premium of the ligneous crops in the area through the implementation of a model that depends on the historical evolution of the insured product, i.e. water availability (Martin *et al.* 2001). In La Campiña, the variables determining water availability for irrigation are the water stock in reservoirs, groundwater and rainfall. However, groundwater levels are difficult to measure because there are no reliable sources of data for the area. The DMP for the GRB uses annual rainfall as a proxy variable to assess the quantitative state of the permeable detritic aquifers of La Campiña (GRBA 2007, 2010). Consequently, our model has two relevant variables: the rainfall and the water stock of reservoirs. We start by estimating the probability density function (PDF) of both variables. Subsequently, the quantity of water available for irrigation in every scenario and its corresponding probability are estimated according to the applicable decision rules, and the potential water savings are estimated. Finally, we use a deterministic agronomic model to estimate the yield of every ligneous crop in La Campiña, its corresponding production value and indemnity and the basic risk premium for every possible scenario.

3.1. Water availability

In La Campiña, water availability is a function of rainfall, piezometric levels and water stock in reservoirs. In the following sections, we calculate the PDF for the water stored in the reservoirs and rainfall (which serves as a proxy variable for piezometric levels) to determine the probability associated with every level of water availability.

3.1.1. Rainfall

Rainfall is the most important source of water in La Campiña agricultural district for two reasons: (i) it provides effective rainfall captured directly by crops; and more importantly, (ii) rainfall recharges the permeable detritic aquifers of the area, which are the main source of water for irrigation. Rainfall is a stochastic variable that can be adjusted to a PDF, which allows for the assignment of a probability ($y = z(p)$) to each level of rainfall, expressed in mm (p). The data used correspond to the period 1941–2008 (MARM 2011). The PDF is calculated with a best-fit gamma function of the following type (McWhorter *et al.* 1966, Martin *et al.* 2001, Gómez Gómez and Pérez Blanco 2012):

$$y = z(p|a,b) = \frac{1}{b^a \Gamma(a)} p^{a-1} \exp\left(\frac{-p}{b}\right) \tag{1}$$

where a and b are the scale and shape parameters, respectively. Table 1 presents the maximum likelihood estimators (MLEs) of the parameters.

3.1.2. Water stock in reservoirs

On the basis of work by Gómez-Ramos *et al.* (2002), Pérez Blanco *et al.* (2011) and Gómez Gómez and Pérez Blanco (2012), we adjust the PDF of the level of water stored in reservoirs at the beginning of the irrigation season using the Weibull function. This function assigns a probability (w) to each stored water level (s), measured as a percentage of the storage capacity, in La Campiña. The data used correspond to the period from 1968 to 2008 (MARM 2008). The Weibull function can be expressed as follows:

$$w = j(s|e,d) = \frac{d}{c}\left(\frac{c}{d}\right)^{d-1} \exp\left(-\left(\frac{s}{c}\right)^{d}\right). \tag{2}$$

Table 2 shows the MLEs of the parameters in the function above.

Table 1. Rainfall Gamma function. The dependent variable is mm of rainfall.

Variable	Coefficient
a (scale)	15.35[a] (2.79)
b (shape)	37.75[a] (3.28)
No. of observations	68

Note: [a]Significant at 1% level. Source: Authors' research from MARM (2011).

Table 2. Surface water stored: Weibull function. The dependent variable is the percentage of dam-stored water over dam storage capacity.

Variable	Coefficient
c (scale)	0.61[a] (0.12)
d (shape)	4.79[a] (0.57)
No. of observations	40

Note: [a]Significant at 1% level. Source: Authors' research from MARM (2008).

3.2. *Decision rules*

At the beginning of every irrigation season, the water authority estimates the quantity of water required for irrigation (*TIR*)[8] according to the crops present in the sub-basin and their historical evapotranspiration rates. Later, the water authority assesses the water availability in the reservoirs and the annual accumulated precipitation (GRBA 2010) to determine the quantity of water to be delivered to agriculture.

Traditionally, the percentage of *TIR* that was effectively satisfied (*TIRr*) followed discretional decision rules. This situation changed with the approval of the DMPs, which clearly established a set of drought thresholds with specific associated restrictions. Nonetheless, DMPs still offer the possibility to apply additional water restrictions, which follow discretional criteria, during exceptional junctures (e.g. during extreme droughts or after a lasting drought to speed up the recovery) (GRBA 2007). Thus, both decision rules are in force.

3.2.1. *Traditional decision rules*

In contrast with the situation created by the recently approved DMP in the GRB, the decision rules followed thus far have been the result of a combination of social agreements, opinions of experts and discretion depending on water availability, with no written rules to be applied in any case. To formalise these decisions, we use the available data on the quantity of water effectively delivered to farmers, measured as a percentage of *TIR* satisfied. The data span a range of 19 years (1989–2008). We found that the relevant variables explaining the percentage of *TIR* satisfied are water stored in reservoirs (measured as a percentage over total storage capacity) (s) and annual rainfall in mm (p). The relationship between the percentage of *TIR* satisfied ($h(p,s)$) and both of these variables are linear (Gómez Ramos *et al.* 2002). The parameters of the function are estimated using ordinary least squares (see Table 3).[9]

3.2.2. *DMP decision rules*

The recently approved DMP of the GRB quantifies the particular situation at hand and the severity of the problem using objective and publicly observable drought thresholds that are dependent on the quantitative state of the groundwater bodies and indirectly assessed through the annual rainfall (p). The plan establishes the following four drought thresholds: (i) when rainfall values are regarded as *normal* ($p < 425$), there are no additional explicit restrictions, and the quantity of water available for irrigation is thus the same as in the traditional rules scenario; (ii) the quantity of water available for irrigation is reduced by 5% ($h = 0.95$) when the amount of rainfall is less than the pre-alert threshold ($325 < p \leq 425$); (iii) if the alert limits are exceeded ($275 < p \leq 325$), the quantity of water available for irrigation is reduced by 30% ($h = 0.7$); and (iv) in emergency situations ($p \leq 275$), the quantity of water that is delivered for irrigation is reduced by 70% ($h = 0.3$) (GRBA 2007).

3.2.3. Combined decision rules

We define $l_{p,s}$ as a discrete water restriction variable whose value depends on the DMP's decision rules (and thus on rainfall), its corresponding h (q for normal, 0.95 for pre-alert, 0.7 for alert and 0.3 for emergency) and the traditional decision rules that apply under exceptional circumstances ($h(p,s)$):

$$l_{p,s} \begin{cases} h(p,s), & \text{if} \quad p > 425 \\ \min\left(h(p,s), 0.95\right), & \text{if} \quad 325 < p \leq 425 \\ \min\left(h(p,s), 0.7\right), & \text{if} \quad 275 < p \leq 325 \\ \min\left(h(p,s), 0.3\right), & \text{if} \quad p \leq 275. \end{cases} \tag{3}$$

Water delivered for irrigation is thus a function of rainfall and the water stock in reservoirs ($TIRr(p,s)$):

$$TIRr(p,s) = l_{p,s} * TIR. \tag{4}$$

3.3. Evapotranspiration satisfied

We measure the expected crop evapotranspiration (ET) for every irrigated ligneous crop in La Campiña, according to the Spanish Ministry of Environment standard method, using data from 1941 to 2009 (MARM 2011).[10] The expected evapotranspiration is partially addressed by effective rainfall. Effective rainfall (ER) is a function of stochastic rainfall and a series of parameters that can be safely assumed to be constant (Cuenca 1989)[11]:

$$ER = g(p). \tag{5}$$

The portion of evapotranspiration (ET) that is not addressed by effective rainfall is the irrigation water requirement (WR):

$$WR = ET - g(p). \tag{6}$$

The WR can either be satisfied through irrigation or left unaccounted for, depending on the available water resources and the decision rules in force. The total quantity of water delivered for irrigation was estimated in the previous section ($TIRr(p,s)$). Nonetheless, only a fraction of the $TIRr(p,s)$ effectively contributes to satisfy evapotranspiration due to water losses during the abstraction, transportation and irrigation stages. The effective irrigation resources ($EIR(p,s)$), or the part of the irrigation resources that effectively satisfy the irrigation water requirements, are a function of $TIRr(p,s)$ and the overall efficiency of the irrigation system (e_s), which is approximately 61% in La Campiña (GRBA 2007):

$$EIR(p,s) = TIRr(p,s) * e_s. \tag{7}$$

The percentage of the evapotranspiration satisfied (%ET) can now be calculated from the previous equations:

$$\%ET_{p,s} = \frac{g(p) + EIR(p,s)}{ET}. \tag{8}$$

Each %ET has an associated probability ($q(p,s)$), which depends on the stock (s) and rainfall (p) values. Using expressions (1) and (2), this probability is expressed as follows:

$$q(p,s) = z(p) * j(s). \tag{9}$$

The expected evapotranspiration satisfaction (E_{ET}), the resulting expected irrigation deficit (ID) and potential groundwater depletion ($PotGW$) are defined as follows:

$$E_{ET} = \int_{p=0}^{\max p} \int_{s=0}^{\max s} [z(p) * g(p) + j(s) * z(p) * (EIR(p,s))] \tag{10}$$

where $\max_p$ and $\max_s$ are the values of the variables p and s that make the cumulative density function equal to 1 (i.e. the probability of any value above this limit is zero).

$$ID = ET - E_{ET} \tag{11}$$

$$PotGW = \frac{ID}{e_s}. \tag{12}$$

3.4. Agronomic production functions and production value

The agronomic production of a given crop depends largely on the percentage of evapotranspiration satisfied. However, making the production function of a crop dependent only on the percentage of evapotranspiration satisfied implies that other variables that may affect the production function, such as soil type, fertilisers and phytosanitaries, climatic variables and others, are excluded. However, if we consider this set of variables to be constant, it is still possible to develop sound and rigorous agronomic production functions that provide results close to the observed values (SCRATS 2005, Pérez Blanco *et al.* 2011). Thus, we obtain the agronomic production in kg ($Q_{p,s}$) as a function of the percentage of evapotranspiration satisfied (%$ET_{p,s}$) and other variables that are assumed to be constant (k):

$$Q_{p,s} = f(\%ET_{p,s}, k). \tag{13}$$

The reference agronomic production functions for the considered crops are obtained after a comprehensive bibliographical review. Subsequently, these functions are adapted to the characteristics of the area of study, if there are not site-specific production functions (SCRATS 2005, MARM 2010). To adapt the production functions, it is assumed that the local characteristics have fixed effects that shift the agronomic production functions but maintain their elasticity and marginal productivity. The resulting production functions are quadratic:

$$Q_{p,s} = a * \%ET^2_{p,s} + b * \%ET_{p,s} + c. \tag{14}$$

Next, we obtain the value of production, which is the product of the agronomic production ($Q_{p,s}$) and the updated average prices of the last 10 years (P) (MARM 2007).

$$V_{p,s} = Q_{p,s} * P. \tag{15}$$

The value of production is the reference value for the calculation of the basic risk premium. Prices are assumed to be constant because neither revenue insurance (price, yield and costs) nor income insurance (price and yield) exist in the EU, where yield insurance prevails. As a result, price variability is not considered in our model.

3.5. *Basic risk premium*

The main element of any insurance market is the estimation of the basic risk premium that, given the likelihood of a catastrophic event, guarantees a certain level of coverage for the insured with no losses for the insurer in the medium-long term. The indemnity conceded by drought insurance in the case of drought losses in the EU is subject to two prerequisites: (i) losses must be institutionally acknowledged; and (ii) losses must be larger than the minimum threshold predetermined by the insurance company, usually as a percentage of the production value.

(1) The drought indemnity is only paid when the relevant authorities formally declare a drought. In the case of La Campiña, the system is considered to suffer a drought when it is subject to at least a pre-alert state (i.e. $p \leq 425$). We generate a dichotomous variable, a_p, to include this condition in our model.

$$\begin{cases} a(p) = 1, & \text{if} \quad p \leq 425 \\ a(p) = 0, & \text{if} \quad p > 425 \end{cases} \tag{16}$$

(3) Additionally, insurance systems only cover at most a percentage of the expected value of the production in a normal hydrological year (V_{exp}). This threshold (μ), which is 70% in Spain (Bielza *et al.* 2008b), aims to reduce the impact of the moral hazard (Miranda 1991). Consequently, the indemnity ($IND(p,s)$) is defined as follows:

$$IND(p,s) = \begin{cases} \mu * V_{\text{exp}} - V_{p,s}, & \text{if} \quad 0 \leq V_{p,s} < \mu * V_{\text{exp}} \\ 0, & \text{if} \quad V_{p,s} \geq \mu * V_{\text{exp}} \end{cases}. \tag{17}$$

The expected indemnity ($IE_{p,s}$) for each crop is calculated as follows:

$$IE_{p,s} = \int_{p=0}^{\max p} \int_{s=0}^{\max s} [z(p) * j(s) + a(p,s) * IND(p,s)] \tag{18}$$

Finally, the basic risk premium ($BRP_{p,s}$) is calculated as the ratio of expected indemnity to expected production in a normal hydrological year ($V_{\exp}$):

$$BRP_{p,s} = \frac{IE_{p,s}}{V_{\exp}}. \tag{19}$$

4. Results

The methodology above has been applied to the La Campiña sub-basin. First, we have estimated the potential of commercial drought insurance to prevent illegal water use during all possible drought events (Table 4).

The new legal and institutional framework after the implementation of the DMPs and the RBMPs results in irrigation restrictions of 6.06 hm^3/year in La Campiña. Without the proper incentives in place (i.e. formal drought insurance), this framework leads to a potential aquifer overexploitation of 9.94 hm^3/year (i.e. informal insurance). This framework results in a deficit of approximately 8% of the total annual renewable resources in La Campiña. This deficit would be much larger during an emergency drought, when expected illegal abstractions would equal 20.5 hm^3/year (16.4% of annual renewable resources in La Campiña), according to our model. It should be noted that this deficit corresponds exclusively to ligneous crops (39.7% of irrigated surface in La Campiña). However, the combination of DMPs and RBMPs with a formal drought insurance system may prevent these illegal water abstractions and thus strengthen the sustainability of the system.

Next, we calculate the expected production value in a normal hydrological year ($V_{\exp}$), the expected indemnity (considering every possible scenario) ($IE_{p,s}$) and the

Table 3. Irrigation resources estimation under traditional decision rules. The dependent variable is the percentage of TIR satisfied in the GRB.

Variable	Coefficient
S	0.35[a] (0.13)
p	0.0007 [a] (0.0002)
R^2	0.89
Adjusted R^2	0.88
No. of observations	19

Note: [a]Significant at 1% level. Source: Authors' research.

Table 4. Expected evapotranspiration satisfaction (E_{ET}), expected irrigation deficit (ID) and expected potential groundwater depletion ($PotGW$) during drought events in absolute terms (hm^3) and as a percentage of ET satisfied (%ET) in La Campiña.

Variable	Value
E_{ET} (hm^3)	36.82
E_{ET} (%ET)	85.9%
ID (hm^3)	6.06
ID (%ET)	14.14%
$PotGW$ (hm^3)	9.94

Source: Authors' research.

Table 5. Expected production value in a normal hydrological year (V_{exp}), Expected Indemnity ($IE_{p,s}$) and Basic Risk Premium ($BRP_{p,s}$) for irrigated ligneous crops in La Campiña.

Variable/crop	*Prunus armeniaca*	*Cerasus*	*Citrus reticulata*	*Malus domestica*	*Citrus × sinensis*	*Olea europaea*	*Vitis*
Irrigated land (ha)	315	4685	2	997	4208	9087	11
V_{exp} (EUR)	1457	6626	11,467	3944	5840	2072	7828
$IE_{p,s}$ (EUR)	33	147	1212	39	597	10	14
$BRP_{p,s}$	2.26%	2.22%	10.57%	0.98%	10.22%	0.50%	0.18%

Source: Authors' research. Reference agronomic production functions were obtained from MARM (2010) (all crops), SCRATS (2005) (citrus trees), Pastor *et al.* (2005) (*Olea europaea*), Almarza (1997) (Vitis) and Pérez Pastor (2001) (*Prunus armeniaca*), Parra *et al.* (2009) (*Malus domestica*).

basic risk premium ($BRP_{r,s,p}$) for the ligneous crops of La Campiña. The values are displayed in Table 5, along with the irrigated surface of every ligneous crop in La Campiña.

Higher basic risk premiums are observed in citrus trees. *Citrus × sinensis* shows a $BRP_{p,s}$ of greater than 10%. This high value may pose a significant challenge for the development of commercial agricultural drought insurance in the region because *Citrus x sinensis* is one of the most representative crops not only in La Campiña (21.8% of total surface) but also in southern and south-eastern Spain. The $BRP_{p,s}$ obtained for *Citrus reticulata*, although consistent with the values obtained for other citrus trees, should not be considered representative because of the small surface covered by these crops in La Campiña. The same can be said for the *Vitis*. *Prunus armeniaca* and *Cerasus* have intermediate drought insurance $BRP_{p,s}$ slightly above 2%. Finally, *Malus domestica* (0.98%) and *Olea europaea* (0.5%), which together represent 52.2% of the irrigated ligneous crops surface, have affordable $BRP_{p,s}$ below 1%.

5. Conclusions

Drought insurance for irrigated agriculture does not exist in Europe. However, the necessary conditions for its development in drought-sensitive areas in Southern Europe are in place after the implementation of RBMPs and DMPs. Our hypothesis is that under the clearer and publicly enforced rules on water abstractions contained in both plans, drought insurance has the potential to reduce illegal groundwater abstractions and stabilise agricultural incomes.

In this paper, we have estimated the cost (i.e. basic risk premium) of this insurance market in the agricultural district of La Campiña (Spain) and the potential water savings that could be attained with the joint implementation of DMPs, RBMPs and drought insurance for irrigation. The results obtained in our case study in La Campiña indicate that the basic risk premium is reasonable and the environmental outcome is significant. Nonetheless, the viability of a private insurance market also depends on other sources of risk that are independent of the insured product, namely, adverse selection, moral hazard (Miranda 1991) and, in the medium-long term, the ability of institutions and private agents to cope with external shocks, such as climate change. Moreover, droughts are a systemic risk in which the risks of the different insured agents are correlated, which may result in catastrophic losses (Bielza *et al.* 2008a, 2008b). These factors demand a certain degree of public initiative (Rejda 2008). In any case, drought insurance for irrigation still offers a better alternative than the baseline scenario, where drought losses are compensated through costly emergency mechanisms that are entirely supported by public institutions and without a water saving target.

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Notes

1. RBMPs are already in effect in every member state, with the exception of Belgium, Greece, Portugal and Spain.
2. Unlike RBMPs, DMPs are not prescriptive, although they are available in several southern European basins in Spain, Italy, Portugal and France, and also in Finland, Netherlands and UK.
3. Only Spain, Italy, Austria and, recently, France have developed drought insurance markets for rain-fed agriculture (Bielza *et al.* 2008b).
4. This figure refers to all of the Andalusian basins, which are the GRB, the Andalusian Mediterranean Basins and the Andalusian Atlantic Basins. The GRB is the largest Andalusian basin and with its 57,527 km^2 covers 59.5% of the total surface of the region.
5. Although alternative insurance methods do exist, such as index financial products or derivatives, they are still in their early stages and are usually experimental (Barnett *et al.* 2005, Bielza *et al.* 2008b).
6. Adverse selection is difficult to clear up in the case of index insurance, although it is easier to resolve for tailored individual insurance plans, as happens in Europe. However, issues of moral hazard in the EU are less important than in other areas because to receive indemnity in the EU, it is necessary to ascertain which event caused the loss, whether the damage affects a sufficiently significant area (that is, that the risk has not affected only one individual farmer) and whether the insured or guaranteed yield can be corrected according to the productive conditions of the insured farm (Bielza *et al.* 2008a). Additionally, a deductible $(1-\mu)$ is applied to discourage this type of behavior (e.g. 15% in France, 20–30% in Italy, 30% in Spain).
7. In most EU countries, including Spain, the insurance market is in the hands of no more than three insurance companies, and this low competition may result in even higher premiums.
8. Spanish river basins estimate *TIR* as the quantity of water required to meet the demand of the 80th percentile of annual historical evapotranspiration, with a global efficiency of the water provisioning system of 60%.
9. For values of *TIR* greater than 100%, the function is truncated and equal to 1.
10. MARM methodology follows a combination of the Thornthwaite and Penman-Monteith Methods (see, for example, Allen *et al.* 2006).
11. Effective rainfall (*ER*) is estimated using the Soil Conservation Service–USDA methodology for Spain (Cuenca 1989), and it is a function of the humidity deficit ($f(D)$), rainfall (p) and evapotranspiration (ET). It is measured in annual mm: $ER = g(p) = f(D) \cdot [1.25\, p^{0.824} - 2.93] \cdot 10^{0.000955 \cdot \mathrm{ET}}$.

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Annex: Summary of variables and parameters

Variable	Description
p	Rainfall (mm)
s	Water stored in reservoirs (percentage over the storage capacity)
$y = z(p\|a,b)$	Gamma PDF
$\mathrm{w} = j(s\|a,b)$	Weibull PDF
a	Scale parameter, Gamma PDF
b	Shape parameter, Gamma PDF
c	Scale parameter, Weibull PDF
d	Shape parameter, Weibull PDF
TIR	Quantity of water required for irrigation according to the crops present in the sub-basin and their historical evapotranspiration data
$h(p,s)$	Percentage of *TIR* satisfied under traditional decision rules
h	Percentage of *TIR* satisfied under the DMP decision rules
$l_{p,s}$	Water restriction variable resulting from the combination of $h(p,s)$ and h
$TIRr$	Percentage of *TIR* satisfied
ET	Expected crop evapotranspiration
$ER = g(p)$	Effective rainfall
WR	Irrigation water requirements
e_s	Overall efficiency of the irrigation system
$EIR(p,s)$	Effective irrigation resources
$\%ET$	Percentage of ET satisfied
$q(p,s)$	Probability of $\%ET$
E_{ET}	Expected evapotranspiration satisfaction
ID	Expected irrigation deficit
$PotGW$	Expected potential groundwater depletion
$Q_{p,s}$	Agronomic production function (kg)
k	Other variables in the production function, which are assumed constant
P	Average prices over 10 years
$V_{p,s}$	Production value
V_{exp}	Expected production value in a normal hydrological year (without drought)
$a(p)$	Dichotomous variable – Drought threshold
μ	Maximum indemnity (% over V_{exp})
$IND(p,s)$	Indemnity
$IE_{p,s}$	Expected indemnity
$BRP_{p,s}$	Basic risk premium

Insuring water: A practical risk management option in water-scarce and drought prone regions?

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Abstract: Recurrent water deficits in various arid and semi-arid Mediterranean basins are largely covered by illegal groundwater abstractions uncontrolled by the water authorities. Aquifers thus play the role of buffer stocks and are used by farmers as a reliable, though informal, insurance system. This has led to continuous groundwater depletion and increased scarcity and drought risk over the last few decades. An effective solution to this problem requires the replacement of this spontaneous, informal and uncoordinated insurance scheme with a formal and planned system that can be coordinated with the objective of reducing overexploitation. In this paper we develop a methodology to estimate the fair risk premium and the potential water savings associated with drought insurance for irrigated agriculture. This method is illustrated with its application to the Campo de Cartagena Agricultural District in the Segura River Basin (Spain). Results show that although the potential for illegal abstractions is high (9.5 hm^3/year), the cost of the insurance system is ten times lower than the amount that risk-averse farmers are willing to pay for water security. This information may serve as the starting point for the design of a drought insurance system able to cope with other relevant institutional challenges.

Keywords: Agriculture, Drought insurance, Drought Management Plan, Groundwater Stochastic models.

JEL classification: Q15, Q18, Q25, Q51, Q58.

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1. INTRODUCTION

Water scarcity is the most pressing environmental issue in EU Mediterranean areas. This situation is to a large extent attributable to irrigated agriculture, which in less than 50 years has doubled its surface and now represents between 70% and 80% of total water use (Massarutto, 2003; EEA, 2009). Although irrigation expansion has significantly enhanced agricultural income, it has also increased the water dependency of the sector and has progressively brought water demand closer to water supply, thus making agriculture more vulnerable to drought. This has generated powerful incentives for illegal water abstractions, which have increased along with drought frequency and intensity (WWF, 2006a; 2006b). Illegal abstractions may threaten the sustainability of the ecological system and are particularly difficult to control, since they are usually located over dependable and uncontrolled groundwater sources (Gómez & Pérez, 2012). As groundwater stocks have been depleted, water scarcity has become chronic and policy makers have called for measures to reduce overexploitation and increase water security.

However, the effectiveness of these measures has been burdened so far by the prevailing paradigm that considers water demand as an exogenous variable outside the field of water policy. A direct consequence of this paradigm is that water policy has been mostly based on expensive *supply oriented policies*, such as the construction of major infrastructure or the modernization of irrigation devices, which paradoxically have ended up increasing water use, reducing water availability and undermining the robustness and resilience of the system and its ability to cope with future droughts. More recently, the high costs of supply policies in a time of crisis and the continuous increase in water use have forced EU authorities to implement *command and control* (C&C) policies. Unlike traditional supply policies, C&C policies introduce clear water abstraction rules to avoid overexploitation during droughts (Drought Management Plans, DMPs) and new coercive mechanisms to enforce these rules (River Basin Management Plans, RBMPs) (CHS, 2008; 2013). However, although these policies increase the likelihood of detecting and penalizing offenders, they do not alter the existing incentives behind illegal water use, which in some irrigated areas may generate profits that exceed the potential costs of being detected. Accordingly, C&C policies may be mostly ineffective in areas where the income gap between irrigated agriculture and rainfed agriculture is large, as is the case in many Mediterranean areas (Mendelsohn & Saher, 2011).

At present, *economic instruments* are gaining momentum as an alternative means of reducing water use. Economic instruments were first introduced into the EU water policy agenda through Article 9 of the Water Framework Directive (WFD), dedicated to water pricing (EC, 2000) Economic instruments can be defined as incentives designed and implemented with the purpose of adapting individual decisions to collectively agreed goals (e.g. the environmental objectives of the WFD). There is a diverse range of economic instruments being applied in different EU member states, including tariffs for water services, trading schemes or voluntary agreements, among others. On the other hand, there are other economic instruments that are still being investigated and for which there is no available *ex-post* evidence on their performance. In these cases, an *ex-ante* assessment is necessary to determine whether or not they are implementable, and if so, if they will produce the desired environmental outcome. This is the case with drought insurance for irrigated agriculture.

The EU has mostly classic, individually tailored agricultural insurance schemes (e.g., yield insurance in the case of drought insurance for *rainfed* agriculture). Although alternative collective insurance schemes such as index-based or financial derivative products have been explored, they have not succeeded so far due to a series of problems[1] (Bielza et al., 2008b). Agricultural insurance in the EU is generally private, although a certain degree of public support is necessary in a number of cases (as with drought insurance). Competition is low and in many countries the market is in the hands of no more than two or three insurance companies, who may operate within a pool and assume the risk in a co-insurance regime. In general, the development of agricultural insurance in the EU member states is heterogeneous and depends on the risk level faced by each country and the financial support for the insurance system by the public sector (Bielza et al., 2008a).

Drought insurance for irrigated agriculture may have the capacity to improve drought management and reduce illegal overexploitation in EU Mediterranean areas. This system guarantees a stable agricultural income during droughts so that the incentives for groundwater depletion are reduced. Accordingly, drought insurance may replace the current scheme in which farmers are transferring their individual risk to water ecosystems and thus to future generations by a formal risk-sharing scheme based on intra-generational and voluntary agreements. However, this mechanism requires a proper institutional set-up in which data on piezometric levels are transparent and up to date, drought indicators are objective, water users are sensitive to the overexploitation problem and drought insurance systems for irrigated agriculture are implementable. While the implementation of the DMPs and RBMPs has served to make relevant progress with the first three preconditions (CHS, 2008; 2013), the latter is still pending since drought insurance for *irrigated agriculture* does not exist in Europe (Bielza et al., 2008a).

The main problem for the development of drought insurance for irrigated agriculture is related to the systemic nature of droughts. Droughts may affect large areas and consequently drought indemnity in profitable irrigated areas may result in unaffordable losses for a conventional insurer. In addition, insurance markets are usually plagued with moral hazard and adverse selection problems, and drought insurance for irrigated agriculture is no exception. Nonetheless, insurance systems have developed strategies to reduce the impact of these problems: moral hazards can be significantly reduced with the establishment of a deductible of the insured product that avoids full loss recovery, and thus the existence of individual strategic behavior; the adverse selection problem can be addressed within the so-called *combined insurance scheme* – a system that offers

[1] Index-based products are best suited for homogeneous areas where all farms have highly correlated yields (for example, in the Corn Belt in the USA). Given the heterogeneity of climates, geography and production systems in many EU countries, the efficiency of index-based products is lower here. In addition, time series of yield losses in the EU are often only available at a regional level, comprising relatively large regions. Some of these regions (like Andalusia or Castile and León in Spain) are large and heterogeneous, making it difficult to create an index that can be used for all farmers in the region; in these cases, the use of yield data at a more disaggregated level would be advisable or even necessary. Finally, there are also some regulatory problems that may make index-based products incompatible with the Community directives (Bielza et al., 2008b).

drought insurance only as a part of a comprehensive insurance package that covers a varied range of agricultural risks (not only those that are more likely to happen) and thus reduces the uncertainty for the insurer; and the systemic risk is tackled with public support[2]. These strategies have made possible the development of drought insurance markets for *rainfed* agriculture in some EU countries (e.g., Austria, France, Italy and Spain) (Gómez et al., 2011).

However, drought insurance for irrigated agriculture has not been developed, and drought losses in irrigated areas in these countries are covered through expensive and publicly supported emergency funds (Meuwissen et al., 2003; Rejda, 2008). It thus seems that the main explanation for the absence of drought insurance for irrigated agriculture in the EU has more to do with insufficient institutional development than with the cost, which would actually be lower for the public sector than with the current system. Indeed, drought insurance for irrigated agriculture is increasingly regarded in some member states as an inexpensive means of compensating farmers and reducing illegal abstractions in an economic context marked by budget cuts. For example, preliminary negotiations between the insuring firms and the Ministry of Agriculture, Food and Environmental Affairs are already ongoing in Spain (Representatives of the Spanish Association of Agrarian Insuring Firms, personal communication)

In such a policy context, assessing the implementability and advancing the potential water savings that can be attained through drought insurance for irrigated agriculture becomes of paramount importance. This paper presents a methodology which combines a stochastic water availability model, a decision model and site-specific agronomic production functions in order to estimate i) the potential water savings and ii) the fair risk premium that would stem from a hypothetical drought insurance market. The *fair risk premium* is the quotient of the expected indemnity (which is equivalent to the share of the yield losses effectively compensated by the insurer[3], measured in constant prices[4]) to the production value in a normal hydrological year (i.e. without drought), and can be interpreted as the minimum long-term cost for this scheme to be provided by a competitive and risk-neutral insurance firm. Thus, it is a crucial value in assessing the financial viability of private drought insurance for irrigated agriculture. It is important to note that the fair risk premium is a first step in the development of drought insurance markets for irrigated agriculture and does not imply that the final risk premium will be close to this value; as stated above, this will depend on the incidence of the moral hazard, adverse selection and systemic risk problems and on the availability of public support.

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[2] For example, in Spain catastrophic losses are covered by the *Insurance Compensation Consortium*. The Consortium assumes cover for the extraordinary risks on a subsidiary basis and will pay indemnification when a private insurer has assumed cover and is subsequently not able to settle claims. The Consortium is funded via a surcharge on insurance policies and a fixed percentage of every premium contracted by insurance companies is paid to the organization.

[3] Insurance systems compensate only a fraction of the yield losses. This avoids full loss recovery and thus reduces the incidence of moral hazard.

[4] Neither income insurance (price, yield and costs) nor revenue insurance (price and yield) exist in the European Union, where yield insurance prevails (Bielza et al., 2008a). As a result price variability is not considered in our model.

This methodology is illustrated by applying it to the irrigated ligneous crops[5] in the Campo de Cartagena Agricultural District in the Segura River Basin (Spain).

The paper is structured as follows: in Section 2, we introduce the area where the case study is applied – the Campo de Cartagena Agricultural District. Section 3 presents the methodology used to estimate the fair premium risk, and Section 4 presents the results obtained. Section 5 discusses the results and concludes the paper.

2. THE CASE STUDY AREA: CAMPO DE CARTAGENA IN THE SEGURA RIVER BASIN (SPAIN)

The Campo de Cartagena Agricultural District is located in the Segura River Basin (SRB) in the south-east of Spain (see Figure 1). The SRB is a good example of the abovementioned trends. In order to reduce water scarcity, supply side policies have been common in the basin. This has included the construction of dams and canals, intense irrigation modernization within the framework of the National Irrigation Plan 2000-2008 and, more importantly, the construction of a massive water transfer in the 1970s with the capacity to transfer up to 600 million cubic meters (or cubic hectometers, hm^3) per year from the Tagus' headwaters to the SRB, known as the Tagus-Segura Water Transfer (CHS, 2013). However, none of these infrastructures delivered the expected outcome, and water use continued to increase and eventually exceeded water supply. More recently, C&C policies have complemented traditional supply policies. As a result, the SRB has already approved its DMP (CHS, 2008) and is about to publish its RBMP, for which there is already a preliminary report available(CHS, 2013).

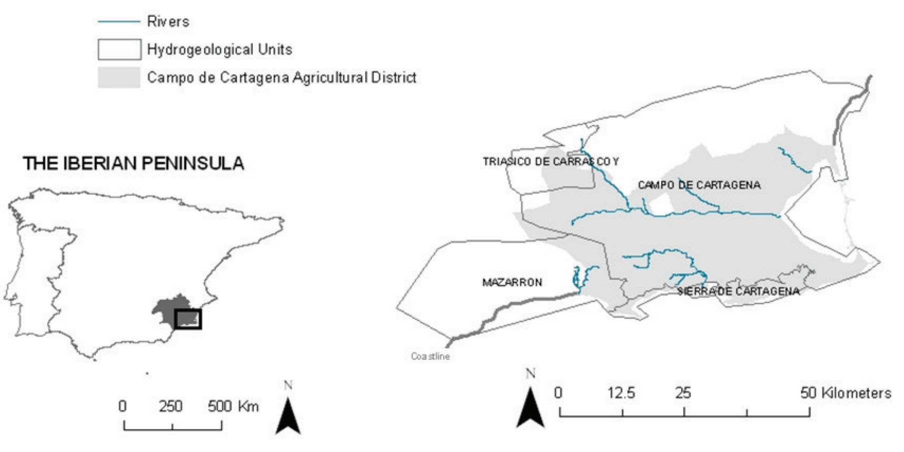
The SRB has traditionally been an overexploited basin. Many hydrogeological units in the basin were declared overexploited in the 1980s (CHS, 2013) and several restrictions to water use have been formally established since. This includes a prohibition on issuing additional water rights for irrigation since 1986 (CHS, 2013). However, between 1990 and 2000 irrigated surface grew at an average rate of 6,500 ha/year, and currently only 155,313 ha of the 225,356 ha under irrigation in the Region of Murcia (comprising 71.4% of the total irrigated land in the SRB) has formal water rights (IDR-UCLM, 2005). At the same time water use for irrigation, which amounts to 89% of overall water use (CHS, 2013), continued growing steadily: in 2003, the ratio between water abstraction and renewable resources in the river basin was an alarming 1.27; by 2009, this ratio had shot up to 2.5, denoting one of the most serious cases of overexploitation in Europe (EEA, 2009; CHS, 2013). In spite of the development of a new regulatory framework based on the SRB's DMP and RBMP, the economic and political cost of enforcing water use rights is recognized as prohibitive given the high income from irrigated lands, which in Campo de Cartagena is 80 times larger than from rainfed agriculture (Maestu et al., 2008). As a result, illegal abstractions have continued and the SRB accumulated groundwater overexploitation now amounts to 7,000 million cubic meters (hm^3), including aquifers whose resources have been exhausted to such a degree that, even in the absence of

[5] Ligneous crops refers to crops from woody plants (i.e. plants that produce wood as their structural tissue) which includes trees and shrubs (fruit and berry trees, bushes, vines, olive trees).

more abstractions, it would take more than a century for them to completely recover (CHS, 2008).

Campo de Cartagena, in the Sistema Cuenca Sub-basin of the SRB, is an agricultural district with approximately 13,000 ha of irrigated ligneous crops (28.9% of the total irrigated land), of which 39% is devoted to citrus trees. Water use for irrigation amounts to 58 hm^3 in a normal hydrological year, of which 16.7 hm^3/year is supplied by the aquifers in the area (Campo de Cartagena, Mazarrón, Sierra de Cartagena and Triásico de Carrascoy, see Figure 1). Since Campo de Cartagena does not possess a stable supply of surface water (rivers in the area are non-perennial), aquifers are severely overexploited and on average 36% of the formal groundwater abstractions are non-renewable (CHS, 2008). In addition, illegal abstractions during drought incidence are widespread, in spite of the high abstraction costs. This is explained by the profitability of local agriculture, which is one of the most productive areas of Spain (Pérez et al., 2011). In this context, the implementation of drought insurance for irrigated agriculture may serve to significantly reduce the cost of controlling illegal abstractions.

Figure 1: Location of the Segura River Basin in the Iberian Peninsula and detail of the Campo de Cartagena Agricultural District.



Source: Authors' elaboration

3. METHODOLOGY

The fair risk premium is the key element in the design of any commercial insurance and is estimated as the ratio between the expected indemnity (a function of the expected yield losses described above) and the expected yield value in a reference year (in this case, a normal hydrological year). The expected indemnity and agricultural production are estimated from an assessment of the historical evolution of the insured product (Martin et al., 2001), in this case water availability, and its impact on the agricultural output through the use of agronomic production functions. We follow the standard procedure which makes the production functions dependent only on water availability, assuming the remaining variables to be constant (Pérez et al., 2011). The methodology that we present below allows the calculation of the expected indemnity and the expected agricultural production, as well as the resultant fair risk premium, through the development of a risk-production model which depends on three stochastic variables (rainfall, runoff and stock), institutional decision rules, site-specific agronomic production functions and a set of agricultural and insurance market variables. The model is made up of five stages:

i) The first stage calculates the amount of water available in different scenarios and its associated probability, which is a function of the three stochastic variables: local rainfall (which satisfies plants' water needs through the effective rainfall directly captured by crops) runoff, and the amount of water stored in the reservoirs in the whole basin (which are used to determine the amount of water delivered to the irrigation system) (CHS, 2008).

ii) The second stage estimates the amount of water delivered to the irrigation system in accordance with runoff levels and stored water available and a set of decision rules (CHS, 2008).

iii) The third stage first obtains the expected evapotranspiration. This value and the results in i) and ii) are used to calculate the percentage of evapotranspiration satisfied in each water availability scenario and the water demand in excess of available resources (irrigation deficit), which gives incentives to engage in illegal abstractions.

iv) The fourth stage develops a deterministic agronomic model that estimates the yield (and yield value, using agricultural market variables) of every crop in each scenario as a function of the percentage of evapotranspiration satisfied obtained in iii).

v) Finally the fair risk premium is estimated as the ratio of the expected drought indemnity to the expected production value, taking into account some special features of the drought insurance markets traditionally used to limit the impact of systemic risk and asymmetric information.

3.1. First Stage: Water availability

In Campo de Cartagena the water authority assigns the irrigation resources accounting for the amount of water stored in the reservoirs of the SRB and the basin's annual runoff (CHS, 2008). Consequently, water availability in Campo de Cartagena is a function of the local rainfall and of the annual runoff and water stored in the reservoirs of the whole basin. Local rainfall is much lower than in other locations within the basin and has a negligible incidence over total runoff or water stock in the SRB due to the downstream location of Campo Cartagena. Thus, we consider rainfall as an independent variable. On the other hand, although the reservoirs of the SRB are partially dependent on the SRB's runoff, a large share of the water stored in the reservoirs comes from external resources transferred from the Tagus' headwaters through the Tagus-Segura Water Transfer, which has the capacity to transfer 600 hm^3/year (as compared with the SRB average runoff of

650 hm^3/year) (CHS, 2013). In turn, the amount of water transferred from the Tagus River Basin (TRB) is a complex decision which depends on a discretionary assessment made by institutions based on the runoff and the water stored in the reservoirs of the TRB (CHS, 2008). In addition, the SRB is an extensively regulated basin and its reservoirs have the capacity to store up to 1,141 hm^3 (1.75 times the average runoff), which means that its surface water stored depends not only on annual runoff and water transfers, but also on the runoff and transfers (and thus on the runoff and surface water stored in the TRB) of preceding years. Consequently, the link between runoff in the SRB and surface stored water in the SRB is weak and we treat both variables as independent.

In the following sections we obtain the probability density functions (PDF) of the three relevant variables (rainfall, runoff and surface water stored) in order to determine the probability associated with each level of water availability.

3.1.1. Rainfall

Rainfall is a stochastic variable which can be adjusted to a PDF. This allows assigning a probability ($y = z(p)$) to each rainfall level (p). Climatological research has supported the use of a gamma distribution to characterize the distribution of climatological variables exhibiting a physical lower bound of zero but no upper bound, such as precipitation or runoff (Martin et al., 2001; [Yue et al., 2001; Scholzel & Friederichs, 2008). Accordingly, we obtain the rainfall PDF as the best fit gamma function of the following type:

$$y = z(p|a,b) = \frac{1}{b^a \Gamma(a)} p^{a-1} \exp(\frac{-p}{b}) \quad [1]$$

where *a* and *b* are, respectively, the scale and the shape parameters. Table 1 presents the maximum likelihood estimators (MLEs) of the parameters. We use rainfall data for the Campo de Cartagena Agricultural District in the period 1941-2008 (AEMET, 2012).

Table 1: Rainfall gamma function. The dependent variable is mm of rainfall per year.

Variable	Coefficient
a (scale)	16.358[a]
	(2.821)
b (shape)	22.9964[a]
	(2.286)
No. of observations	68

Estimated by maximum likelihood. Standard errors in parentheses.
a: significant at 1 the per cent level.
Source: Authors' elaboration from MARM, 2009b

3.1.2. Runoff

Annual runoff in our model is measured as a percentage over the storage capacity of the reservoirs in the river basin. As with rainfall, we adjust the runoff to a gamma PDF (see discussion above). This allows assigning a probability ($q = f(r|a,b)$) to each runoff level (r). The gamma function can be represented as follows:

$$q = f(r|a,b) = \frac{1}{b^a \Gamma(a)} r^{a-1} \exp(\frac{-r}{b}) \quad [2]$$

where *a* and *b* are, respectively, the scale and the shape parameters. Table 2 shows the best fit parameters for the runoff function. We use runoff data for the SRB in the period 1941-2008 (MARM, 2008).

Table 2: Annual runoff gamma function. The dependent variable is the percentage of annual runoff over the total surface water storage capacity.

Variable	Coefficient
a (scale)	6.1813[a] (1.088)
b (shape)	0.1143[a] (0.012)
No. of observations	68

Estimated by maximum likelihood. Standard errors in parentheses.
a: significant at the 1 per cent level.
Source: Authors' elaboration from MARM, 2008

3.1.3. Water stored in reservoirs

Surface water stored is usually closely linked to runoff and rainfall patterns, and thus could also be adjusted using a gamma PDF (Scholzel & Friederichs, 2008). However, this is not the case in interconnected and extensively regulated basins, where the link between surface water stored and runoff is weak. Spanish south-eastern river basins like the Segura and Andalusian Mediterranean River Basins and some areas of the Guadalquivir and Júcar river basins are a good example of this type of water management. In these cases, the literature supports the use of Weibull functions for the adjustment of surface stored water PDFs. (Gómez-Ramos et al., 2002; Pérez et al., 2011; Gómez & Pérez, 2012). A Weibull function assigns a probability (w) to each amount of surface water stored (s), measured as a percentage over the dam storage capacity (DSC) of the SRB. The Weibull function can be represented as follows:

$$w = j(s|a,b) = \frac{b}{a}\left(\frac{a}{b}\right)^{b-1} \exp\left(-\left(\frac{s}{a}\right)^{b}\right) \quad [3]$$

Table 3 shows the MLEs of the parameters in the function above. We use data on the water levels of the reservoirs in the SRB during the period 1941-2008 (MARM, 2008). These data are then aggregated to obtain a synthetic index of the total surface water stored as a percentage over the DSC of the SRB dams, which is the dependent variable in the function above:

Table 3: Surface water stored: Weibull function. The dependent variable is the percentage of dam-stored water over dam storage capacity (DSC).

Variable	Coefficient
a (scale)	0.3411[a]
	(0.063)
b (shape)	4.1286[a]
	(0.497)
No. of observations	68

Estimated maximum likelihood. Standard errors in parentheses.
a: significant at the 1 per cent level.
Source: Authors' elaboration from MARM, 2008.

3.2. Decision rules

At the beginning of every irrigation season, the water authority estimates the amount of water required for irrigation (TIR)[6] according to the crops present in the sub-basin and their historical evapotranspiration data. Then, the water authority assesses annual runoff and water availability in the reservoirs (CHS, 2008) and determines the percentage of TIR that will be effectively satisfied (h).

Traditionally, the percentage of TIR effectively satisfied has followed discretional decision rules. This situation changed with the approval of the DMPs, which clearly establish a set of drought thresholds with specific restrictions associated. Nonetheless, DMPs still offer the possibility to follow discretional criteria during exceptional junctures (e.g. during extreme droughts or after a lasting drought, to speed up the recovery) (CHS, 2013), so actually both decision rules are in force.

3.2.1. Traditional decision rules to determine water delivery for irrigation

In contrast with the situation created by the recently approved DMPs, the decision rules followed until now have been the result of a combination of social agreements, opinions of expert judges and discretion, with no written rules to be applied in any case, depending on the water available for the crop season. To formalize these decisions, we use the available data on the amount of water effectively delivered to farmers measured as a percentage of TIR satisfied. The available data span a range of 15 years (1992 to 2007) (CHS, 2008). We found that the only relevant variable explaining the percentage of TIR satisfied in the past has been the runoff (r). The relationship between the percentage of TIR satisfied (h) and runoff (r) is linear ($h = h(r)$) (Gómez-Ramos et al., 2002). The parameters of the function are estimated using ordinary least squares[7].

[6] Spanish river basins estimate *TIR* as the amount of water required to cover the 80th percentile of annual historical evapotranspiration with a global efficiency of the water provisioning system of 60%.

[7] For values of TIR over 100%, the function is truncated and equals 1.

Table 4: Irrigation resources estimation under the traditional decision rules. The dependent variable is a percentage of *TIR* conceded in the SRB.

Variable	Coefficient
Runoff (percentage over dam storage capacity)	1.351[a] (.131)
R2	89.14
Adjusted R2	88.31
No. of observations	15

Estimated by ordinary least squares. Standard errors in parentheses.
a: significant at 1 the per cent level.
Source: Authors' elaboration from CHS (2010b)

3.2.2. DMP decision rules over water for irrigation

The recently approved DMP for the SRB quantifies the particular situation at hand and the severity of the problem by using an objective and publicly observable drought index dependent on the values of the annual runoff and stock ($I_e(r,s)$). The drought index is calculated as follows (CHS, 2013):

$$I_e = \begin{cases} \frac{1}{2}\left(1+\frac{B_i-B_{med}}{B_{max}-B_{min}}\right), if\ B_i \geq B_{med} \\ \frac{1}{2}\left(\frac{B_i-B_{min}}{B_{med}-B_{min}}\right), if\ B_i < B_{med} \end{cases} \qquad [4]$$

where B_i is an indicator that is unique for each sub-basin. In *Sistema Cuenca*, Campo de Cartagena's corresponding sub-basin, B_i is obtained as follows:

$$B_i = \frac{2*DSC*r+DSC*s}{3} \qquad [5]$$

where r is the annual runoff as a percentage of the total dam storage capacity (DSC) and s is water stock in reservoirs as a percentage of the total DSC. Using r and s maximum, minimum and average values during the reference period, we obtain B_{max}, B_{min} and B_{med}, respectively.

The DMP establishes the following four drought thresholds: i) when water stored levels are regarded as *normal* ($I_e > 0.5$), there are no explicit restrictions, and thus the percentage of TIR effectively satisfied (h) is the same as in the baseline or traditional decision rules scenario ($h = h(r)$); ii) water for irrigation is reduced by 10% ($h = 0.9$) when available water falls below the pre-alert threshold ($0.35 < I_e \leq 0.5$); iii) if the alert limits are exceeded ($0.2 < I_e \leq 0.35$), water for irrigation is reduced by at least 25% ($h = 0.75$); and iv) in emergency situations ($I_e \leq 0.2$), water for irrigation is halved ($h = 0.5$) (CHS, 2008).

3.2.3. Combined decision rules

We define $l_{r,s}$ as a discrete water restriction variable whose value depends on the drought index (and thus on runoff and surface water stored values):

$$l_{r,s} = \begin{cases} \min(h(r), 0.5), if\ I_e \leq 0.2 \\ \min(h(r), 0.75), if\ 0.2 < I_e \leq 0.35 \\ \min(h(r), 0.9), if\ 0.35 < I_e \leq 0.5 \\ h(r), if\ I_e > 0.5 \end{cases} \quad [6]$$

Water delivered for irrigation is thus a function of runoff and water stored in reservoirs ($TIRr(r, s)$):

$$TIRr(r, s) = l_{r,s} * TIR \quad [7]$$

3.3. Third stage: Evapotranspiration satisfaction, irrigation deficit and illegal abstractions

We measure the expected crop evapotranspiration (ET, in mm) for every irrigated ligneous crop in La Campiña according to the Spanish Ministry of Environment standard method, using data for the period 1941-2009 (MARM, 2011)[8]. The evapotranspiration thus obtained is partially covered by effective rainfall (ER, in mm). ER is a function of stochastic rainfall (p), whose PDF was obtained in [1], and a series of parameters which can be safely assumed to be constant[9]:

$$ER = g(p) \quad [8]$$

The part of evapotranspiration (ET) that is not covered by effective rainfall is the irrigation water requirements (WR, in mm):

$$WR = ET - g(p) \quad [9]$$

WR can either be satisfied with irrigation or left uncovered, depending on the available water resources and the decision rules in force. The total amount of water delivered for irrigation was obtained in the previous section ($TIRr(r, s)$). Nonetheless, only a fraction of the $TIRr(r, s)$ effectively contributes to satisfying evapotranspiration due to water losses during the abstraction, transportation and irrigation processes. The effective irrigation resources ($EIR(r, s)$), or the part of the irrigation resources that effectively satisfy evapotranspiration, is a function of $TIRr(r, s)$ and the overall efficiency of the irrigation system (e_{sys}), which is around 87% in Campo de Cartagena (CHS, 2008):

$$EIR(r, s) = TIRr(r, s) * e_{sys} \quad [10]$$

The percentage of the evapotranspiration satisfied ($\%ET$) for *a random year* with a given rainfall (p), runoff (r) and water stored in reservoirs (s) can now be obtained from the previous equations, as follows:

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[8] MARM methodology follows a combination of the Thornthwaite and Penman-Monteith Methods (see, for example, Allen et al., 2006).

[9] Effective rainfall (*ER*) is estimated using the Soil Conservation Service–USDA methodology for Spain (Cuenca, 1989), which is a function of the humidity deficit (*f(D)*) (whose value for the SRB can be taken from Cuenca (1989)), rainfall (*p*) and expected evapotranspiration (*ET*) (constant). It is measured in annual mm:

$$ER = g(p)$$

$$\%ET_{r,s,p} = \frac{g(p)+EIR(r,s)}{ET} \quad [11]$$

Each $\%ET$ value has an associated probability ($u(r,s,p)$), which depends on the stock (*s*), runoff (*r*) and rainfall (*p*) values for that year. Using expressions [1], [2] and [3] this probability can be expressed as follows:

$$u(r,s,p) = u(r \cap s \cap p) = f(r) * z(p) * j(s) \quad [12]$$

Finally, the expected evapotranspiration satisfaction (E_{ET}) and the resultant expected irrigation deficit (ID) and expected potential for illegal groundwater abstractions ($PotGW$) are defined as follows:

$$E_{ET} = \int_{r=0}^{max_r} \int_{p=0}^{max_p} \int_{s=0}^{max_s} [z(p) * g(p) + f(r) * j(s) * EIR(r,s)] \quad [13]$$

$$ID = ET - E_{ET} \quad [14]$$

$$PotGW = \frac{ID}{e_{gw}} \quad [15]$$

Where max_r, max_p and max_s are the values of the variables r, p and s that make the cumulative density function equal to 1 (i.e. the probability of any value above this limit is zero), e_{gw} is the efficiency of illegal groundwater abstractions in the SRB, estimated at 25% (CHS, 2008).

3.4. Fourth stage: Agronomic production functions and production value

The agronomic production of a given crop depends largely on available water, either from rainfall or irrigation ($\%ET_{r,s,p}$). However, making the production function of a crop dependent only on the evapotranspiration satisfied suggests the exclusion of other variables that may affect the production function (soil type, fertilizers and phytosanitaries, climatic variables, etc.). On the other hand if we consider this set of variables to be constant (k) it is still possible to develop sound and rigorous agronomic production functions that provide results close to observed values (SCRATS, 2005; Pérez et al., 2011). Thus we obtain the agronomic production in kg/ha ($Q_{r,s,p}$):

$$Q_{r,s,p} = f(\%ET_{r,s,p}, k) \quad [16]$$

The reference agronomic production functions for the crops considered are obtained after a comprehensive bibliographical review (Almarza, 1997; Pérez-Pastor, 2001; SCRATS, 2005; Alarcón et al., 2006; Mañas et al., 2007; Vivas Cacho, 2010). In the cases where there are no site-specific production functions available, production functions are adapted to the characteristics of the area of the case study (SCRATS, 2005; MARM, 2010). To do so it is assumed that the local characteristics have fixed effects that shift the reference agronomic production functions but maintain their elasticity and marginal productivity. The resultant production functions are quadratic:

$$Q_{r,s,p} = x_1 * \%ET_{r,s,p}{}^2 + x_2 * \%ET_{r,s,p} + x_3(k) \quad [17]$$

Where $\%ET_{r,s,p} \in [0,1]$. x_1 and x_2 are the parameters that determine the impact of water availability over the agronomic production and $x_3(k)$ is a constant that captures the effect that the remaining variables (k, assumed constant) have over the agronomic production. For all the crops it is considered that the agronomic production is null when

$\%ET_{r,s,p}$ falls below 50% (SCRATS, 2005)[10]. This value is increased to 70% for citrus trees and 85% for *Pyrus communis* and *Prunus persica*. Then we obtain the value of the production, which is the product of agronomic production ($Q_{r,s,p}$) times the updated average prices of the last 10 years (P)[11] (MARM, 2007).

$$V_{r,s,p} = Q_{r,s,p} * P \quad [19]$$

During a normal hydrological year without drought (i.e. when $\%ET_{r,s,p} = 100\%$), we refer to the agronomic production as *normal agronomic production* (Q_{norm}). Accordingly, the production value in a normal hydrological year is denominated as the *normal production value* (V_{norm}).

3.5. Fifth stage: Fair risk premium

The key element of any insurance market is the estimation of the fair risk premium that, given the likelihood of a catastrophic event, guarantees a certain level of coverage for the insured with no losses for the insurer in the medium-long term. The fair risk premium is thus obtained as the quotient of the expected indemnity (IE) to the production value in a normal hydrological year without drought (V_{norm}). The latter has already been obtained in the previous section; in what follows, we estimate the expected indemnity for drought insurance in irrigated agriculture.

The indemnity conceded by a drought insurance system in the EU after a drought is subject to two prerequisites:

i) First, the drought must be institutionally acknowledged. This is of crucial importance for private insurers, as financial public support is only available when a drought is institutionally declared. In Spain, a drought is officially declared when the DMP comes into force (i.e., there is a prealert, alert or emergency state) and irrigation restrictions are implemented. In the particular case of the SRB, this happens when $I_e \leq 0.5$. We generate a dichotomous variable, $t_{r,s}$, to include this condition in our model:

$$\begin{cases} t(r,s) = 1, if\ I_e \leq 0.5 \\ t(r,s) = 0, if\ I_e > 0.5 \end{cases} \quad [20]$$

ii) Second, insurance systems cover at most a fraction of the yield losses, in order to avoid moral hazard. This means that a deductible (μ) applies. In

[10] Actually, there is no data on production values for any species for $\%ET_{r,s,p}$ below 50% (70% for citrus trees). Below this value, water availability is not enough for agricultural production, and may even put at risk the survival of ligneous crops (SCRATS, 2005). Moreover, DMPs consider ligneous crops a priority use (only after household and environmental supply) and guarantee a minimum water supply to avoid catastrophic losses. As a result of this, once calibrated the quadratic functions of these species may show a negative agronomic production under extreme droughts. We rule out this possibility and for production values < 0, the function is truncated and equals 0.

[11] In the EU, drought insurance systems insure only yield losses, excluding price variability. As a result, the yield/production value is obtained using constant prices. In the case of Spain, prices are assumed to be an average of the previous years (Bielza et al., 2008a;2008b; Gómez et al., 2011).

Spain this deductible is 30%, implying that a maximum of 70% $(1-\mu)$ of the yield losses would be compensated by the insurer (Bielza, 2008b). For example, in an extreme situation in which all the yield is lost ($V_{r,s,p}=0$), the indemnity would equal 70% of the normal production value ($(1-\mu)*V_{norm}$) and thus 70% of the yield losses would be recovered. However, if there have been yield losses due to a drought ($V_{r,s,p}<V_{norm}$) but the observed production value is still greater than or equal to this maximum compensation threshold ($V_{r,s,p}\geq(1-\mu)*V_{norm}$), the indemnity would be zero (0% loss recovery). Finally, if the production value is between 0 and $(1-\mu)*V_{norm}$, the indemnity would equal the maximum possible indemnity minus the observed production value ($(1-\mu)*V_{norm}-V_{r,s,p}$), meaning that the loss recovery ratio would be in the interval (0%, 70%). Indemnity in every possible scenario ($IND(r,s,p)$) is thus defined as follows:

$$IND(r,s,p)=\begin{cases}(1-\mu)*V_{norm}\ ,if\ V_{r,s,p}=0\\(1-\mu)*V_{norm}-V_{r,s,p},if\ 0<V_{r,s,p}<(1-\mu)*V_{norm}\\0\ ,if\ V_{r,s,p}\geq(1-\mu)*V_{norm}\end{cases}\quad [21]$$

Now we use the equations [20] and [21] and the PDFs obtained in Section 3.1 (equations [1], [2] and [3]) to obtain the expected indemnity (IE) for each crop:

$$IE=\int_{r=0}^{max_r}\int_{p=0}^{max_p}\int_{s=0}^{max_s}[z(p)*f(r)*j(s)*a(r,s)*IND(r,s,p)]\quad [22]$$

where again max_r, max_p and max_s are the values of the variables r, p and s that make the cumulative density function equal to 1, and $z(p)$, $f(r)$ and $j(s)$ are the probabilities for every value of p, r and s, respectively.

Finally, the fair risk premium (FRP) is obtained as follows:

$$FRP=\frac{IE}{V_{norm}}\quad [23]$$

4. RESULTS

The methodology above has been applied to the particular case of the Campo de Cartagena Agricultural District. First, we estimated the expected evapotranspiration satisfaction (E_{ET}) and the subsequent expected irrigation deficit (ID) and expected potential for illegal groundwater abstractions ($PotGW$). We estimated expected evapotranspiration satisfaction at 43.3 hm^3/year, 92.3% of the total evapotranspiration of 45.7 hm^3/year. Accordingly, the expected irrigation deficit amounts to 2.4 hm^3/year, which given the low efficiency of illegal groundwater abstractions results in an expected potential for illegal groundwater abstractions of 9.5 hm^3/year (more than half of annual *legal* groundwater abstractions, estimated at 16.7 hm^3). It is important to note that this is an *expected* value: for example, during emergency situations ($I_e\leq 0.2$) the expected potential for illegal groundwater abstractions soars up to 38.8 hm^3/year according to our model (while in normal hydrological years it is 0).

$PotGW$ can be interpreted as the potential of drought insurance to prevent illegal groundwater use during drought. Accordingly, a successful drought insurance system can help to both stabilize farmers' income and significantly improve the quantitative status of groundwater bodies. However, the viability of this system needs to be assessed first. This viability depends on the long-term cost of the drought insurance for the insurer (FRP). In order to estimate the FRP we start by calibrating the parameters of the agronomic production functions for the ligneous crops in the Campo de Cartagena Agricultural District.

Table 5: Agronomic production functions, ligneous crops in Campo de Cartagena Agricultural District (kg/ha)

Crop/Coeff.	x_1	x_2	$x_3(k)$	R^2
Prunus dulcis	-7,796.7 [a]	15,609 [a]	1,346.7 [a]	0.89
	(364.2)	(1680.6)	(160.9)	
Prunus armeniaca	6,224.1 [a]	52.41 [a]	8,933.4 [a]	0.81
	(859.5)	(6.5)	(1395.3)	
Citrus × limon	-16,967 [a]	53,265 [a]	-13,288 [a]	0.99
	(2282.6)	(5874.9)	(1601.2)	
Citrus reticulata	-13,712 [a]	49,445 [a]	-12,335 [a]	0.92
	(1650.9)	(5005.5)	(583.4)	
Prunus persica	-61,794 [a]	110,955 [a]	-24,804 [a]	0.86
	(6571.4)	(13261.9)	(3314.4)	
Citrus × sinensis	-16,013 [a]	52,947 [a]	-13,208 [a]	0.93
	(2148.4)	(6266.0)	(1485.3)	
Pyrus communis	-43,034 [a]	88,101 [a]	-25,626 [a]	0.79
	(4392.6)	(9480.1)	(3886.1)	
Vitis	-11,918 [a]	23,859 [a]	2,058.4 [a]	0.83
	(562.8)	(3452.5)	(313.6)	

Estimated by OLC. Standard errors in parentheses.
a: significant at the 1 per cent level.

Sources: Authors' elaboration from MARM (2010) (all crops), SCRATS (2005) (citrus trees), Mañas et al. (2007) (Prunus dulcis), Almarza (1997) (Vitis), Alarcón et al. (2006) (Prunus persica), Vivas Cacho (2010) (Pyrus communis) and Pérez Pastor (2001) (Prunus armeniaca).

Once the agronomic production functions have been calibrated, we apply the methodology above to obtain an estimation of the long-term cost of this scheme, both in absolute terms (expected indemnity, IE) and as a percentage over the production value in a normal hydrological year (fair risk premium, FRP). The table below shows these results and also the intermediate values of the agronomic production (Q_{norm}) and the production value (V_{norm}) in a normal hydrological year. Results are displayed for every ligneous crop in the Campo de Cartagena Agricultural District:

Table 6: Normal Agronomic Production (Q_{norm}), Normal Production Value (V_{norm}), Expected Indemnity (IE) and Fair Risk Premium (FRP) for the ligneous crops in Campo de Cartagena Agricultural District.

Variable/Crop	*Prunus dulcis*	*Prunus armeniaca*	*Citrus × limon*	*Citrus reticulata*	*Prunus persica*	*Citrus × sinensis*	*Pyrus communis*	*Vitis*
Q_{norm} (kg/ha/year)	9 159	15 210	23 010	23 398	25 001	23 726	19 441	13 999
V_{norm} (EUR/ha/year)	5 428	5 286	5 825	2 559	9 630	2 351	3 775	2 313
IE (EUR/ha/year)	0.5	49.7	213.2	233.6	13.5	199.4	5.3	0.2
FRP (%)	0.01%	0.94%	3.66%	9.13%	0.14%	8.48%	0.14%	0.01%

Source: Authors' elaboration

Citrus trees have the highest $FRPs$: 9.1% for the *Citrus reticulata*, 8.5% for the *Citrus × sinensis* and 3.7% for the *Citrus × limon*. This is largely explained by their comparatively high drought vulnerability and the resultant high IE (which ranges between 199 and 234 EUR/ha/year), which makes these crops the most expensive to insure. These results are particularly important as citrus trees are the most significant ligneous crops in Campo de Cartagena, representing 11% of the total irrigated surface and 39% of the surface of irrigated ligneous crops.

The remaining fruit trees show lower $FRPs$. *Prunus persica* has the highest V_{norm}, but a low vulnerability to drought and thus a low IE, which results in a FRP of only 0.1%. *Prunus armeniaca* has a lower but still high V_{norm}, a small IE and a FRP of 0.9%. *Pyrus communis* has a particularly low IE and a FRP of 0.1%[12].

Finally, the lowest FRP is that of the *Vitis* and the *Prunus dulcis* (0.01%). *Vitis* and *Prunus dulcis* have traditionally been rainfed crops in the SRB and this explains their higher drought resilience (IE for the *Prunus dulcis* and *Vitis* is 0.5 and 0.2 EUR/ha/year, respectively) and their low FRP.

The low FRP of crops like *Pyrus communis* does not mean that they are not vulnerable to suffering drought losses; rather, this means that the largest losses appear in droughts with a low probability (very low local rainfall and low runoff and water stored

[12] Although both *Pyrus communis* and *Prunus persica* need a high $\%ET_{r,s,p}$ (above 80%) in order to have a positive production, the total evapotranspiration of these crops (ET) is low and therefore the irrigation water requirements (WR) are low or even null during years with sufficient rainfall. As a result, the likelihood of suffering the impacts of a drought are more reduced than with crops with a higher ET, such as citrus trees.

simultaneously). In any case, low FRP crops are still vulnerable to moderate losses during likely droughts, and insurance can help to stabilize agricultural income. Nonetheless, the relatively low evapotranspiration of low FRP crops implies that their potential to reduce illegal groundwater abstractions is limited (e.g. the irrigation water demand of *Vitis* is 47% of that of *Citrus x limon*). In comparison, the production value of water-intensive crops with high FRP, such as citrus trees, may drop as much as 100% during likely droughts.

All the *FRP*s obtained are at least ten times lower than the amount farmers in Southern Spain are willing to pay for water security, as estimated by Gutiérrez-Martín & Gómez (2011). This is consistent with economic theory, as there is considerable evidence showing that farmers are risk-averse individuals who are ready to pay in excess of their expected loss in order to have a more secure income (Torkamani & Haji-Rahimi, 2001; Binici et al., 2003; Tobarra & Castro, 2011).

5. DISCUSSION AND CONCLUSION

Water overexploitation, particularly from agriculture, is the most important environmental threat faced by EU Mediterranean areas. This problem is delaying the fulfillment of the environmental goals prescribed by Community law and is reducing or even stopping household water supply during drought occurrence, which are both priority objectives of the Water Framework Directive (EC, 2000). The authorities have tried to address this problem with the systematic implementation of supply and C&C policies. The most significant drawback of these policies is that they do not change the powerful incentives that drive water demand. For example, supply side policies that increase the availability of water resources have often been perceived by users as an opportunity to increase agricultural income (by shifting from rainfed to irrigated agriculture, for example) rather than as a chance to increase water security through the improvement of the quantitative status of overexploited water bodies. Consequently, these policies have backfired and have ended up reinforcing the trajectory of the system towards higher water scarcity and more frequent and intense droughts.

An effective response to the problem of water overexploitation needs to put in place the necessary instruments to orientate individual voluntary choices towards the collectively agreed goal of improving the status of water bodies. In the particular case of illegal water abstractions in profitable irrigated areas, drought insurance may constitute a powerful economic instrument to reduce water use and transfer the cost of water security to intensive water users at an acceptable cost. In our case study area the implementation of drought insurance schemes would have the potential to save up to 9.5 hm^3/year on average at a basic cost of less than 10% of the production value in a normal year.

Although drought insurance for irrigated agriculture does not exist in Europe, the necessary conditions for its development in drought-sensitive areas of Southern Europe may be in place with the new Community legal framework, characterized by the DMPs and the RBMPs. RBMPs and DMPs indicate a clear commitment to stop illegal abstractions during drought occurrence through an improved surveillance mechanism (which includes an up-to-date groundwater inventory), clear abstraction rules and more rigorous application of sanctions. However, the effectiveness of these plans has been limited so far due to the incentives to engage in illegal groundwater abstractions during drought as a result of the high agricultural income. Drought insurance systems for irrigated agriculture can stabilize agricultural income and therefore present relevant synergies with DMPs and RBMPs that may be used to reduce the incentives for illegal abstraction.

However, even if the right *packaging* is found, this does not guarantee its success. There is a vast amount of literature that shows how past institutional choices influence the cost of changing institutions and may eventually block the adoption of a policy, even if this policy is desirable (institutional *lock-in*, North, 1990). According to this literature, the transition towards the implementation of an instrument such as drought insurance may be as relevant as the very design of the instrument. There are two key variables in this process: *transaction costs* and *sequencing* (McCann et al., 2005; Garrick et al., 2013).

Transaction costs are particularly relevant in the case of insurance markets. The most relevant include moral hazard, adverse selection and systemic risk. Assessing the impact of transaction costs over the drought insurance premium is beyond the scope of this paper, and further research is needed in this direction. In any case, the $FRPs$ obtained with our model are shown to be at least ten times lower than the amount of money that farmers in Southern Spain would be willing to pay for water security (Gutiérrez-Martín & Gómez, 2011). In principle, this difference leaves enough room to deal with the costs stemming from asymmetric information problems (moral hazard and adverse selection) and systemic risk, as suggested by recent research (estimations of transaction costs for a variety of agricultural insurance markets can be found for example in World Bank, 2005; Bryla & Syroka, 2007; Dick, 2007; and Breustedt et al., 2008).

Sequencing is based on the concept of adaptive efficiency (e.g. Carey & Sunding, 2001). The objective is to select the sequence of institutional innovations with the highest potential to reduce the implementation costs of a given policy over time. These institutional innovations must be designed with a double purpose: i) to reduce the direct transaction costs through cumulative processes of demonstration, learning by doing, etc. in order to enhance the acceptability of the scheme such that the transaction costs of the next institutional change are reduced (as the policy becomes more socially acceptable); ii) the gradual tuning of the instrument so that its effectiveness is increased by the progressive improvement of its design.

One central element in this process is the opportunity to assess how the instrument performs and its contribution to the objectives of the water policy. In this sense it is not only important to produce evidence on, for instance, the number of farmers using drought insurance for irrigated agriculture, but to show that this instrument contributes to reducing scarcity through the better quantitative status of groundwater bodies. This is an essential requirement to show that the scheme is beneficial not just for those directly involved, and to make third parties aware of the advantages of proceeding in this way.

If these obstacles can be successfully addressed, drought insurance may become a useful means of reducing illegal groundwater abstractions during droughts. In fact, drought insurance for irrigated agriculture is increasingly regarded in some member states as an inexpensive means (when compared to expensive emergency funds, Meuwissen et al., 2003) of reducing illegal abstractions in an economic context marked by budget cuts, with preliminary negotiations already ongoing between the public and private sectors in Spain

(Representatives of the Spanish Association of Agrarian Insuring Firms, personal communication).

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Appendix I: Summary of variables and parameters.

Variable	Description
p	Rainfall (mm)
r	Runoff (percentage over the storage capacity)
s	Water stored in reservoirs (percentage over the storage capacity)
$y = z(p\|a, b)$	Gamma PDF (Rainfall)
$q = f(r\|a, b)$	Gamma PDF (Runoff)
$w = j(s\|a, b)$	Weibull PDF
a	Scale parameter, Gamma/Weibull PDF
b	Shape parameter, Gamma/Weibull PDF
TIR	Amount of water required for irrigation according to the crops present in the sub-basin and their historical evapotranspiration data
$h(r)$	Percentage of TIR satisfied under traditional decision rules
h	Percentage of TIR satisfied under the DMP decision rules
$l_{r,s}$	Water restriction variable resulting from the combination of $h(r)$ and h
$TIRr(r, s)$	Percentage of TIR satisfied
ET	Expected crop evapotranspiration
$ER = g(p)$	Effective rainfall
WR	Irrigation water requirements
e_{sys}	Overall efficiency of the irrigation system
$EIR(r, s)$	Effective irrigation resources
$\%ET$	Percentage of ET satisfied
$u(r, s, p)$	Probability of $\%ET$
E_{ET}	Expected evapotranspiration satisfaction
ID	Expected irrigation deficit
e_{gw}	Efficiency of illegal groundwater abstractions
$PotGW$	Expected potential for illegal groundwater abstractions
$Q_{r,s,p}$	Agronomic production function (kg/ha/year)
k	Other variables in the production function, assumed constant
P	Average prices, 10 years
$V_{r,s,p}$	Production value (EUR/ha/year)
Q_{norm}	Expected production in a normal hydrological year (without drought) (kg/ha/year)
V_{norm}	Expected production value in a normal hydrological year (without drought) (EUR/ha/year)
$a(r, s)$	Dichotomous variable – Drought threshold
µ	Deductible (30% in our case study area)
$IND(r, s, p)$	Indemnity
IE	Expected indemnity (EUR/ha/year)
FRP	Fair risk premium (%)

5.3 Water pricing

According to the Article 9 of the WFD: “Member States shall take account of the principle of recovery of the costs of water services, including environmental and resource costs […]” (EC, 2000). However, while water pricing has been extensively used as a financial instrument (i.e., to recover the investment and maintenance costs of water infrastructures), the recovery of environmental and resource costs (i.e., its role as an economic instrument) has been largely neglected. As a result, current water pricing policies do not reflect the actual cost of the resource and do not provide the “[…] adequate incentives for users to use water resources efficiently, and thereby contribute to the environmental objectives of this directive” (i.e., the WFD) (EC, 2000).

As a result of this narrow interpretation, most of the water pricing instruments that have been implemented so far in the EU focus exclusively on the recovery of the financial costs of water supply. This is the case of the most emblematic “environmental” taxes in Spain, which in reality are tariffs designed in most of the cases to recover the costs of sanitation and wastewater treatment infrastructures built to comply with EU standards (EC, 1998, 1991). Some of these pricing schemes include the Tax on water treatment (region of Castile-La Mancha, NUTS2: ES42), Tax on coastal wastewater discharge (Andalusia -ES61-, Murcia -ES62), Tax on radioactive waste (Andalusia), Tax on water (in the Balearic Islands -ES53-, Navarra -ES22-, Valencia -ES52), Tax on water and water pollution (Navarra, Aragón -ES24- and Cantabria -ES13), Tax on certain activities that cause environmental harm (Castile-La Mancha), Tax on the environmental damage caused by some uses of water from reservoirs (Galicia -ES11) and Tax on wastewater treatment (Madrid -ES30) (EEA&OECD, 2013).

Although narrowly focused on financial costs, water pricing has rapidly become a very widely used tool in water policy and this offers a good opportunity to extend its use to environmental and resource costs. In the case of Spanish agriculture, water pricing basically consists of three parts: the regulation fee (*canon de regulación*, covering the abstraction and storage costs of surface water), the water use tariff *(tarifa de utilización del agua*, covering the transportation costs of surface water) and some quotas to pay the expenses of the irrigation communities[23]. For groundwater only the latter exists, since the financial costs of supplying water are assumed by the users (Maestu and Villar, 2007). This complex system generates a wide variety of water prices over the different basins. For example, average water prices in the Segura River Basin (0.061 €/m^3) are 1.6 times larger than those of the Tagus River Basin (0.038 €/m^3) (SRBA, 2013; TRBA, 2008). It is necessary to underline that this gap does not reflect the higher environmental and resource costs of water in water stressed Mediterranean areas, but rather the higher financial costs of abstracting and distributing the water in the Segura River Basin.

Water pricing in Spain has failed to recover the environmental and resource costs, which are particularly relevant in the case of agricultural water use (EEA, 2013). Moreover, financial cost recovery is below 100% in all the Spanish basins (it ranges between 50% and 90% for the abstraction and storage costs and between 54% and 98% for the transportation costs)[24] (Maestu and Villar, 2007). Therefore, in these areas the mere implementation of the full cost recovery principle for financial costs may encourage a more sustainable water use (Caswell et al., 1990; EC, 2000).

Traditionally, it has been assumed that higher prices do reduce water demand. Nonetheless, there is a very intense discussion ongoing regarding the capacity of water pricing to actually achieve this objective. According to some authors (Caswell et al., 1990; Kampas et al., 2012; Rivers and Groves, 2013), whose views are close to those of EU institutions (EC, 2012, 2000; EEA, 2013), higher water prices do reduce agricultural water demand. These authors state that water pricing has the highest potential to balance water demand and supply and may serve as a mechanism to achieve the river basin closure. Moreover, water pricing systems can be designed to prevent aquifer overexploitation where metering is available. In

[23] This payment can adopt different forms from one irrigation community to the other and from one area to the other. In traditional irrigation communities that use surface water, users make an annual payment according to the surface under irrigation. In more water stressed irrigation communities, a dual payment system applies: a payment for the surface under irrigation and a payment for the time during which the land is being irrigated. In new irrigated areas and irrigation communities with mixed water sources (surface and groundwater) a dual payment system applies as well. This is also the case in irrigation communities that rely mostly on groundwater, since the most relevant cost is that of the energy needed to pump the resource. Only where more efficient irrigation systems are in place (drip irrigation) a volumetric system applies (Maestu and Villar, 2007).

[24] In addition, on May 2012 the Spanish government approved the Royal Decree-Law 17/2012 on urgent measures regarding the environment. The most relevant modifications are in the articles 1.3 and 4, which include a clause creating exceptions to the application of the cost recovery principle for water (BOE, 2012).

conclusion, always according to these authors, water pricing contributes to make the economy more resilient by increasing buffer stocks and reducing conflicts around water. However, other authors state that water pricing not only does not reduce agricultural water demand (due to the large gap between the price and value of irrigation water) (Hellegers and Perry, 2006; Perry, 2005), but also may end up increasing water consumption through the incentives to adopt more efficient irrigation technologies that reduce return flows and aquifer seepage (see the *Hydrological Paradox* in Chapter 3) (Medellín-Azuara et al., 2012).

If any evidence is to be extracted from the literature, this would be that both positions may be right and that the impact of water pricing over water use is case study sensitive. In any case, what this debate clearly points out is that water pricing policies should not be regarded as a *panacea*. A thorough *ex-ante* assessment is necessary prior to the implementation of this instrument in order to control for any possible rebound effect in water consumption. Although a null or positive effect of higher prices over water demand would not affect the role of water pricing as a cost recovery instrument, it would demand the implementation of alternative policies to attain the environmental goals of water policy. For example, in areas where an idle capacity of non-conventional water resources is available (e.g., desalinated water), water pricing can be used to make these resources more attractive to farmers and promote the substitution of overused and (financially) cheap conventional resources by (financially) expensive and underused desalinated water.

In the following paper, prepared by the doctoral candidate and the Professors Gonzalo Delacámara and Carlos Mario Gómez Gómez, we develop a revealed preferences model that is used to assess the impact of water pricing over water demand in the Segura River Basin in Spain. The paper is entitled *Water pricing and water saving in agriculture. Insights from a Revealed Preferences Model in a Mediterranean basin*, and is under second review in *Agricultural Water Management*.

Water pricing and water saving in agriculture. Insights from a Revealed Preference Model in a Mediterranean basin

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Abstract: The large irrigation expansion of the last 50 years has significantly increased pressures over water resources in EU Mediterranean basins. Under these circumstances water pricing has been especially encouraged, since within the EU policy context it is widely believed that higher water prices reduce water demand. This paper presents a Revealed Preference Model that provides a clear intuition of the logic behind farmers' choices by using standard economic analysis and implementing a multi-attribute utility function. The model is calibrated for the Agricultural Districts (ADs) in the overexploited Segura River Basin (SRB) in southeastern Spain. Results show that in highly profitable ADs such as those of the SRB, farmers may react to higher prices by reducing their gross margin instead of reducing water use (or employment). These results support the implementation of more ambitious cost-recovery policies and, where possible, the replacement of overexploited conventional resources by more expensive and mostly unused non-conventional resources.

Keywords: Water pricing, Irrigation, Mediterranean basins, Revealed Preference Models

JEL classification: Q15, Q18, Q25, Q51, Q58.

1. INTRODUCTION

The large irrigation expansion of the last 50 years has increased the pressures over water resources worldwide. In Southern Europe, irrigated land has doubled its area during this period and now represents between 70% and 80% of the total water use (Bosello and Shechter, 2013; EEA, 2009). Some basins have been declared overexploited, and this has reportedly been aggravated by more recurrent and intense droughts as a result of climate change (Bosello and Shechter, 2013; Jenkins, 2013; EC, 2008).

This water crisis has led EU institutions to consider different mechanisms to save water in the agricultural sector. In particular, water pricing has gained special momentum during the last years. Article 9 of the Water Framework Directive (WFD) states: "[…] water-pricing policies provide adequate incentives for users to use water resources efficiently, and thereby contribute to the environmental objectives of this directive" (EC, 2000).

The assertion that higher water prices can *per se* reduce water use was already supported in the seminal works by Caswell et al. (1990), Dinar and Subramanian (1997) and Tsur and Dinar (1997), among others. More recently, this negative relationship between prices and water use can also be found for example in Balali et al. (2011), Kampas et al. (2012), Rivers and Groves (2013) and Ward and Pulido-Velazquez (2008), and from an institutional perspective in EEA (2013), Bogaert et al. (2012) and EC (2012). However, there is also a significant body of literature that contends that this assumption is at best debatable (Cornish and Perry, 2003; Cornish et al., 2004; Hellegers and Perry, 2006; Molle, 2001; Perry, 2005; Steenbergen et al., 2007). Using empirical data from different agricultural areas around the world, these authors show that the large gap between the price and value of irrigation water demands a significant increase in water prices in order to attain relevant water savings, this resulting in substantial socio-economic losses in turn.

In addition, recent evidence shows that higher water prices may put in place the necessary incentives to invest in the modernization of irrigation systems to reduce water use per output unit (Medellín-Azuara et al., 2012; Perry, 2011). However, after an increase in irrigation efficiency, although water use and withdrawals may actually fall, water availability for other uses may drop through increased consumptive use[1] (either through a rise of the irrigated area or through a more water intensive crop portfolio), reduced return flows and lost aquifer seepage, in what is known as the *hydrological paradox* (Jensen, 2007; Rodríguez-Díaz et al., 2012; Scheierling et al., 2006; Ward and Pulido-Velazquez, 2008). As a result, water bodies may end up with higher exhaustion levels than before the water price was increased to "save" water.

[1] Water withdrawals refer to water removed from its source for a specific use, while water use is the amount of water demanded by users. The two flows are not the same because of leaks. Finally, water is consumed when a part of the water evaporates or becomes contaminated (Kohli et al., 2010).

Accordingly, although much work has been done on the economics of irrigation water pricing, there is still a remarkable lack of understanding of what impacts can be realistically expected from water pricing policies. This paper intends to shed light on this debate and applies a Revealed Preference Model (RPM) to explore the effects of water pricing over agricultural water demand and consumption and the potential of this policy for water savings and/or conservation. RPMs offer an edge on the traditional agricultural decision models used to date (i.e., linear programming, positive mathematical programming and multi-criteria decision models), such as multi-attribute and non-linear utility functions, more flexibility, a sound calibration mechanism and in general a stronger coherence with basic economic principles (see Section 3 for a more in-depth discussion on this). The model is calibrated for each of the 12 Agricultural Districts (ADs) in the Segura River Basin (SRB) in Spain. After the calibration process, a simulation is run in which the price of conventional water sources (groundwater and surface water) is progressively increased. After several iterations, the water demand curve for each AD is estimated. Lastly, these results are aggregated in order to obtain the water demand curve for the entire SRB District.

It is important to note that the ratio of water demand/use over water consumption remains constant along the curve, since the high irrigation efficiency in the area discourages further investments to the irrigation systems in place[2]. Results show that water demand in the SRB is highly inelastic, mainly as a result of the high gross margin characteristic of irrigated areas in southeastern Spain, which absorbs most of the price shock. Thus, the potential of water pricing *per se* to induce water savings and/or conservation in the agricultural sector in this area is very limited.

However, water pricing can be used to promote the swap of overused and cheap conventional resources by more expensive and underused desalinated water.

In addition, the inelastic water demand and the large gross margins observed in agriculture may serve as an argument in favour of more ambitious cost-recovery policies.

This paper is structured as follows: section 2 introduces the area where the case study is applied, the Segura River Basin in Spain, one of the most overexploited and profitable basins in the EU (EEA, 2009). Section 3 presents the steps that we follow in the calibration process of the RPM and how the calibration errors are estimated. Section 4 shows and discusses the results obtained and aggregates them into a single water demand curve for the whole SRB. Section 5 concludes.

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[2] Water scarcity, recurrent droughts, high expected income and government subsidies have already pushed irrigation efficiency in the SRB to levels above 90% in many ADs (SRBA, 2013). At this point, shifting to more efficient devices without government subsidies may not be profitable due to the higher operation costs (e.g., energy demands) and more complex management practices (Corominas, 2010; Gutierrez-Martin and Gomez, 2011).

2. WATER USE AND WATER PRICING IN THE SEGURA RIVER BASIN (SRB), SPAIN

The SRB is located in southeastern Spain, comprising most of the territory of the Region of Murcia (NUTS2: ES62, according to ISO 3166-2) and parts of Castile-La Mancha (ES42), Andalusia (ES61) and Comunidad Valenciana (ES52). This area benefits from several competitive advantages, including abundant and cheap land[3] and labor[4], adequate solar radiation[5] and proximity to high demand markets.

All these factors make the SRB a thriving agricultural area that shows some of the highest productivity values in Spain and in the EU (Pérez-Blanco et al., 2011). Agricultural land covers 679,976 ha (52.1% of the total area of the basin), of which 269,022 ha are irrigated (20.6% of the total area and 39.6% of the agricultural land). The SRB comprises 12 ADs or *comarcas* (i.e., shires), namely, Sierra Segura and Hellín (in the Castile-La Mancha Region), Vinalopó and Meridional (in the Comunidad Valenciana Region), Nordeste, Noroeste, Centro, Río Segura, Suroeste-Valle Guadalenti and Campo de Cartagena (in the Murcia Region), Vélez and Bajo Almanzora (in Andalusia) (see Figure 1)[6]. It should be noted that these ADs are highly heterogeneous: those located in the coastal areas have the most profitable crops and demand more water than those in upstream areas, where less water intensive and rainfed crops are grown.

Competitive advantages of the SRB explain the significant irrigation expansion witnessed in the area during the last 50 years (Eurostat, 2013). However, water in the SRB is scarce. Eventually, renewable water resources were unable to meet the increasing water demand, and the SRB became an overexploited basin. Many hydrogeological units in the basin were already declared overexploited in the 1980s and several restrictions to water use have been formally established since. This included the prohibition to issue additional water rights for irrigation since 1986 (BOE, 1986). However, since 1990 irrigated surface

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[3] Spain has 261,000 km^2 of agricultural land, the largest in the EU only after France; this represents 52.9% of the total area, as compared to the EU average of 43% (Eurostat, 2013).

[4] Due both to the local labour cost and an elastic labour supply fed for many years from immigration, average gross annual earnings in Spain are 26,568, only slightly above the EU-27 average of 25,942 and well below the Eurozone average of 30,462 (Eurostat, 2013).

[5] Spain has 2,910 sunshine hours per year, while national averages of other member states – with the exception of Portugal, are below 2,500 (FAO, 2013).

[6] Four ADs located within the limits of the SRB have been excluded of our study since they represented less than 1% of the agricultural water use. These ADs are Sierra Alcaraz, Centro and Almansa (in the Castile-La Mancha Region) and Sierra Segura (the part located in the Andalusia Region).

has grown at an average rate of 6,500 ha/year (WWF, 2006), and this has boosted agricultural water use, which currently represents 89% of the total water demand of 1,900 million cubic meters (or cubic hectometers, hm^3) in the basin. Since renewable water resources in the basin equal 760 million cubic meters (SRBA, 2013), there is a large water deficit that is mostly covered through the overexploitation (in some cases, through illegal withdrawals) of the water stock stored in the SRB's aquifers (Gómez and Pérez-Blanco, 2012).

The traditional response of water authorities to water scarcity in Spain has mostly consisted in supply-side policies to increase water availability. In the SRB in particular, this entailed subsidies to drill new wells, the construction and modernization of transportation, distribution and irrigation networks and the building of the Tagus-Segura Water Transfer (TSWT), a major diversion project with the to transfer up to 1,000 million cubic meters/year from the Tagus River Basin located 242 km away[7].

Although these policies have made new irrigation developments possible that have helped to invigorate the local economy and to retain population in rural areas, they have also caused severe environmental problems, such as aquifer depletion and the destruction of riverine ecosystems (e.g., the formerly perennial Segura River currently does not reach the Mediterranean Sea during most of the year).

When it was clear that conventional water sources were already at their limit, authorities turned to non-conventional water sources, including treated wastewater and, especially, desalinated water. Only in the last decade, public authorities invested more than €400 million in the construction and modernization of desalination plants in the SRB (GWI, 2012). In an effort to maintain the pace of infrastructure investment, the Spanish Ministry of Agriculture, Food and Environment is now trying to negotiate an additional €700-million loan, following a €500 million loan used to bailout the public water utility in charge of supplying desalinated water in southeastern Spain (*Acuamed*) in 2012 (GWI, 2013). All this investment and rising energy prices have made desalinated water an expensive source with a production cost around 1€/m^3 (Maestu and Villar, 2007). In spite of the subsidies to make this water source more attractive to farmers (bulk desalinated water is sold in many agricultural areas at 0.36€/m^3) (GWI, 2012), the low when not null price of conventional water sources make desalinated water unattractive (in the SRB, conventional bulk water prices range from 0 €/m^3 in irrigated areas supplied with groundwater to 0.22 €/m^3 in those areas receiving water from the TSWT) (SRBA, 2013).

[7] Although the actual capacity of the TSWT is 1 000 million cubic meters/year, it has been limited to 600 million cubic meters/year by law. However, since its opening in 1978, this infrastructure has been working much below this legal limit and has transferred in average 329.3 million cubic meters/year (SRBA, 2013). In addition, it has been the cause of a major conflict between the regions of Castile-La Mancha (largely belonging to the Tagus River Basin District) and Murcia (largely belonging to the Segura River Basin District).

Consequently, desalinated water is mostly used as a buffer stock during drought events, and only in those areas without access to reliable groundwater sources. As a result desalination plants, with the capacity to supply up to 1/6 of the annual water demand, are being used below 20% of their capacity (i.e., they are supplying 1/30 of the annual water demand) (SRBA, 2013).

On top of that, this water policy has generated unrealistic expectations on the capacity of the system to absorb additional pressures. As a result, all these investments have paradoxically ended up increasing water demand, reducing water availability due to aquifer depletion and undermining the robustness and resiliency of the system and its ability to cope with future droughts. Only in the last decade, the ratio between water abstraction and renewable resources in the SRB has nearly doubled: in 2003, it was an upsetting 1.27; by 2013, this ratio had hit up to 2.5, denoting one of the most serious cases of overexploitation in Europe (EEA, 2009; SRBA, 2013).

The WFD explicitly states that water pricing has to be used as an incentive to adapt water demand to the EU environmental standards, especially in overexploited areas such as the SRB (EC, 2000). Higher prices for conventional water sources in agriculture may improve the status of water bodies in the SRB in two ways: i) they can reduce the expected income and thus constrain water demand from low productive crops; and ii) they favour the substitution of the overexploited conventional water sources by the largely idle non-conventional water sources.

Although the average bulk water price for agriculture in the SRB is the highest in Spain (0.096 €/m^3 for conventional water sources, almost twice as large as the Spanish average of 0.05 €/m^3) (Maestu and Villar, 2007; SRBA, 2013), this price only reflects the higher financial cost of supplying water in the SRB as compared to other basins in Spain (mainly due to the large infrastructure investments and maintenance costs). Therefore, this water price does not take into account the economic value of the resource, including (among others) the environmental costs of water supply, which could significantly increase water price[8].

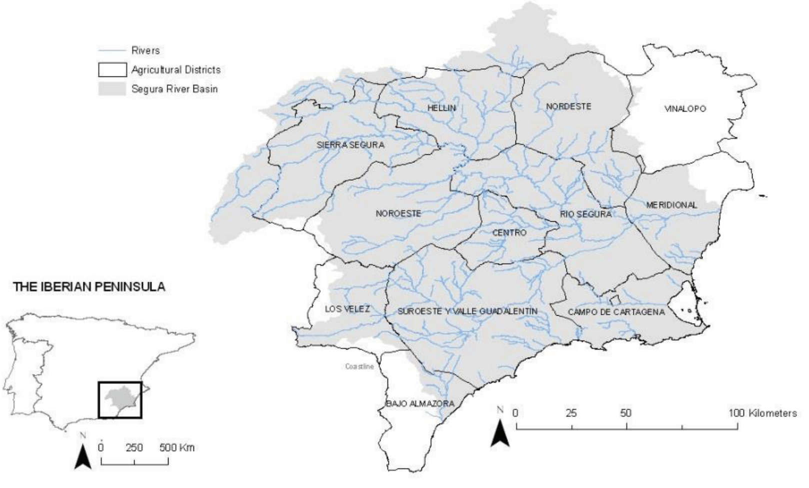
Moreover, the observed water price is not even enough to guarantee a financial full cost recovery, with cost recovery ratios ranging between 54.08% (for own surface water resources) and 80.82% (for the TSWT) (Maestu and Villar, 2007). This is even more shocking once it is considered that most of these investments were ultimately aimed at

[8] According to the Article 9 of the WFD: "Member States shall take account of the principle of recovery of the costs of water services, including environmental and resource costs […]" (EC, 2000). However, while water pricing has been extensively used as a financial instrument (i.e., to recover the construction and maintenance costs of water infrastructures), the recovery of environmental and resource costs has been largely neglected. This can be extended to most EU river basins.

guaranteeing water security in agriculture, a private activity[9]. This alone could justify a price increase based on the grounds of the cost recovery rationale. However, such a policy may also have adverse effects over the local economy, which heavily relies on agriculture. In the Region of Murcia, comprising 71.4% of the total irrigated land in the SRB, agriculture represents 4.9% of GDP and 10.36% of employment. The latter figure has special relevance for a region that holds an unemployment rate of 30.4% (INE, 2013).

Consequently, an appropriate evaluation of the impact of water pricing has to consider not only the effects over total water use, but also the implications of this policy over the economy as a whole. In the next section we develop a RPM that, once calibrated, allows for a comprehensive assessment of the trade-off between environmental and productive uses of water.

Figure 1. Location of the Segura River Basin and detail of the Agricultural Districts



Source: Own elaboration

[9] Water demand from priority uses, namely, environmental flows and household supply, represents less than 10% of the total water demand in the Segura River Basin and could be met with less than 25% of the renewable resources available in the basin (SRBA, 2013). Since water supply for priority uses is guaranteed by law (SRBA, 2008), investments to increase water security were actually aimed at reducing uncertainty in other sectors, and especially in agriculture (89% of the total demand).

3. METHODOLOGY

Economic literature has placed much effort in the last decade to the *ex-ante* assessment of the impact of water pricing over water demand and water consumption. The most commonly used methods so far have been based on Linear Programming (LP), Positive Mathematical Programming (PMP) and Multi-Criteria Decision Methods (MCDM). However, in spite of being widely accepted, these methods are not exempt of some degree of criticism.

The need to represent complex decision problems with limited information has fostered the use of Linear Programming (LP) and Positive Mathematical Programming (PMP) to simulate farmers' response against water pricing and to elicit water demand functions.

Although the origin of LP dates back to the 1950s, it has been widely used in recent years to assess water pricing policies due to its low data requirements and flexibility (see for example Dono et al., 2010 and Kampas et al., 2012). However, this method has been strongly criticized as a result of its failure to approximate, even roughly, realized farm production plans and, therefore, to become a useful methodology for policy analysis (Paris, 2011). This criticism is grounded on the linear nature of this method, which often results in overspecialization and corner solutions.

In addition, LP might be criticized by the way it deals with the parameter specification problem: there is an infinite set of parameters and functions able to lead the model to a perfect calibration, and each set of parameters and functions leads to a different behaviour in response to changing economic prices and policy constraints.

PMP came as a response to the above-mentioned critiques. PMP offers many advantages over LP, including full calibration, a significant reduction in the number of resource, technical, economic and policy constraints, and the use of nonlinear cost functions that guarantee smooth simulation results.

The use of these models to simulate farmers' behaviour and to obtain water demand functions can be found for example in Blanco-Gutiérrez et al. (2011), De Frahan et al. (2007) and Heckelei and Britz (2005)). The general idea of these models consists in using information contained in dual variables of the calibration constraints to bind the solution of the linear profit-maximizing problem to observed activity levels[10]. Once these dual variables are identified they are used to specify a non-linear objective function, such as the production cost, provided that the marginal cost of the activities is equal to its price

[10] This linear model maximizes the profit associated to a vector of activity levels (*x*, represented by surfaces dedicated to a set of crops), with prices and unitary costs considered as constant and subject to a set of resource constraints.

in observed activity levels. This guarantees that both the profit maximization and cost minimization problems simultaneously lead to an optimal solution that exactly matches baseline activity levels (Heckelei and Britz, 2005; Howitt, 1995; Paris and Howitt, 1998)[11].

Although effective, this calibration mechanism is not rooted on explicit economic principles, which makes this the main criticism against PMP. The analyst using PMP might be forced to use *ad-hoc* arguments to explain empirical results. PMP methods do not provide information about estimation errors making uncertainty analysis somewhat unfeasible.

Finally, Multi-Criteria Decision Methods (MCDM) have also played a major role in the assessment of water pricing policies (Rodrigues et al., 2013; Chung and Lee, 2009; Gómez-Limón and Riesgo, 2004; Berbel and Gómez-Limón, 2000). Contrary to PMP methods, in MCDM farmers do not act simply as profit maximizing agents; instead of that, agents consider other relevant attributes in their decision. Therefore, MCDM assume that farmers' preferences can be represented by a weighted sum of different criteria, such as expected profits, risk, management issues and/or others, which provides a better explanation of current decisions. Although this method has succeeded in reproducing the baseline decision, the assumption that farmers respond with linear preferences to changes in policy is again an issue prone to discussion.

Therefore, the construction of water pricing simulation models has been confronted so far with a trade-off between the model's capability to provide numerical results for policy evaluation and its coherence with basic economic principles. However, it is still possible to develop a methodology that is consistent with these principles and yields useful results for policy analysis through the use of Revealed Preference Models (RPM). These applied models provide a clearer intuition of the logic behind farmers' decisions by using standard economic analysis and by implementing a multi-attribute utility function. Moreover, RPM do not need to assume linear preferences (as in LP and MCDM) or implicit costs functions that are not observable (as in PMP). Although the complex programming and optimization procedure and the high data requirements of these models have made difficult their use as a policy assessment and project analysis tool, the advances in computational methods of the last two decades and the recent proliferation of high quality agricultural microeconomic databases in several EU countries make their implementation feasible.

3.1. The Revealed Preference Model (RPM)

[11] The dual variables, obtained in the first stage and used to build the nonlinear objective function in the second, are assumed to capture any type of aggregation or model specification bias, any kind of risk attitude or price expectation as well as any lack of data or data measurement error

This section presents a RPM able to calibrate observed decisions with a procedure rooted in basic microeconomic theory. This method not only allows to obtain simulation results but also offers a clear interpretation of farmers' responses to changing incentives and resource and policy environments. In this model, agents (the ADs of the SRB) decide on their crop portfolio trying to maximize their utility, which is a function of a set of relevant attributes that may contain expected profit, risk avoidance, management complexities and/or others. It is assumed that the explanation of any decision, consisting in a distribution of the available land among the different crop options, relies on an underlying utility function formed by the many attributes that agents use to assess all the alternatives they have, given crop prices and costs, resource availability and the other relevant economic, agronomic and policy constraints. According to that, the observed decisions respond to a decision problem as follows:

$$\underset{x}{\mathrm{Max}}\, U(x) = U(z_1(x);\ z_2(x); z_3(x) \dots z_m(x)) \quad [1]$$

$$\text{s.t.:} \quad 0 \le x_i \le 1 \quad [2]$$

$$\textstyle\sum_{i=1}^{n} x_i = 1 \quad [3]$$

$$X \in F(x) \quad [4]$$

$$z = z(x) \in R^m \quad [5]$$

Where $x \in R^n$ is the decision profile or the crop portfolio (a vector), showing one way to allocate land among crops, and each x_i measures the share of land devoted to the crop i, including a reservation option (x_n) consisting of rainfed agriculture. From the agent's perspective any particular crop may be considered as an asset with a known present cost and an uncertain value in the future (as crop yields are not known in advance). As the available land is taken as given, this investment may be represented as a share (x_i) of available land. $F(x)$ represents the space of feasible decision profiles, given the different constraints[12]: policy, economic, agronomic and environmental. Finally z_i, or alternatively the vector z, are the attributes that farmers value. For example, farmers might prefer decisions with high expected profits, highly predictable yields and prices and not too many managing actions besides planting and harvesting. To accept taking high-risk options, risk averse farmers will ask for a compensation, for example, higher expected profits, and the same can be said about the willingness to accept crop decisions that demand additional management skills.

Let us assume that the observed decision profile and the whole set of constraints defining the feasible decision set are known. Also assume that the set of potentially relevant

[12] These constraints vary for each AD. In our model we consider the following: land availability, available water resources, agricultural vocation (crops that have not been planted in an area before cannot appear in that area in the short run), crop rotation, CAP restrictions and ligneous crops restrictions (the surface of ligneous crops cannot change significantly in the short run).

decision attributes such as, for example, the expected profit, the variance of the expected profit, the hired labour demanded, the cost of inputs over the total cost and all the variables that might be relevant from the farmers' point of view are measurable.

Therefore, the first problem that it is necessary to deal with in order to reveal farmers' preferences is to know which among the potentially relevant attributes are actually relevant to explain the observed decision. The method to answer this question consists in assuming that the relevant set of attributes is the one to which the observed decision is closest to the attributes possibility frontier. In real situations this efficiency frontier cannot be analytically defined with a closed mathematical function and the only way to represent it is by using numerical methods. One practical solution consists in drawing a line from the origin ordinate and through the observed decision attributes and ranging them as far as possible in the space of feasible attributes. This way we can measure the distance from the observed attributes to the efficiency frontier attributes. This procedure can be repeated for any set of potentially relevant attributes and the best candidate to reveal farmers' preferences will be the one whose observed values were closest to its associated efficiency frontier. Formally, this problem must be solved for every member of the Power set (P(z), which comprises all possible combinations of potentially relevant attributes for the farmer) and for its associated observed attributes in the Power set ($P(z_o)$) [13].

The solution to this problem is an application assigning a distance φ_l $(l = 1, \dots, 2^m)$ to each member of the power set $P(z)$. Each member of the power set (i.e., each possible combination of potentially relevant attributes) is denoted by $\tau(x)$, and its associated observed attributes by $\tau_o(x)$. The relevant set of attributes (τ^*) will be the one with the lower distance to the efficiency frontier measured by the parameter $(\varphi - 1)$. Summing up, the preference-eliciting problem can be presented as:

$$\underset{\tau}{\text{Min}}\ \varphi_l - 1 \qquad [6]$$

Where: $\varphi_l = \text{ArgMax}\left[(\varphi)\ \text{s.t.}\ \tau(x) = \varphi(\tau_o(x));\ 0 \le x_i \le 1;\ \sum_{k=1}^{n} x_k = 1;\ X \in F(x); \text{for all}\ \tau \epsilon P(z)\right]$ [7]

$$l = (1 \dots . 2^m) \qquad [8]$$

By solving this problem the set of attributes that better explains current farmers' decisions $(\tau^*)$is obtained. Among the many factors that might be of relevance in farmers preferences, this set of attributes is the one that takes the observed decision closer to the attributes efficiency frontier.

Once a farmer's decision is shown as close as possible to the efficiency frontier, the second problem consists in eliciting the farmers' preferences that explain the observed

[13] A power set P(Z) is the set of all the 2^m subsets of Z and the power set $P_0(Z)$ is the set formed by the 2^m subsets of the numerical set of observed attributes.

decision as a utility maximizing choice. Taking into account the relevant decision attributes obtained at the calibration stage, the multi-attribute utility function is the one that is able to represent farmers' preferences in such a way that the observed decision becomes the optimal choice. Using basic economic principles and knowing the efficiency frontier in the surroundings of the observed decision allows one to integrate such a utility function. Rational decisions imply that in equilibrium farmers' marginal Willingness To Pay in order to improve one attribute with respect to any other is equal to the marginal opportunity cost of this attribute with respect to the other. In other words, the marginal transformation relationship between any pair of attributes over the efficiency frontier ($\mathrm{MTR}_{\mathrm{kp}}$) is equal in equilibrium to the marginal substitution relationship between the same pair of attributes over the indifference curve tangent to the observed decision ($\mathrm{MSR}_{\mathrm{kp}}$).

Now we obtain the relative opportunity cost of each one of the relevant attributes with respect to the others. This opportunity cost is measured by the marginal transformation relationship between any pair of attributes ($\beta_{\mathrm{kp}} = \mathrm{MTR}_{\mathrm{kp}} = \mathrm{MSR}_{\mathrm{kp}}$). This value can be numerically obtained by solving partial optimization problems in the proximity of the observed decision (as for example, searching by how much expected profits would need to be reduced in order to have 1% less uncertainty or, equivalently, what is the maximum expected profit attainable with a slightly lower risk level). The numerical results of the marginal relationship of transformation of any pair of attributes in a reference point over the efficiency frontier (β_{kp}) are the basic information to integrate the farmers' utility function. Provided that farmers behave rationally, in equilibrium, the value (β_{kp}) representing the relative opportunity cost of any attribute in terms of any other is equal to the marginal substitution relationship between the same pair of attributes (which represents the farmers' marginal WTP for an improvement of a given attribute in terms of any other). In other words, in equilibrium, decisions over crop surfaces are such that:

$$\beta_{\mathrm{kp}} = \mathrm{MTR}_{\mathrm{kp}} = \mathrm{MSR}_{\mathrm{kp}} = -\frac{\partial \mathrm{U}/\partial \mathrm{z_p}}{\partial \mathrm{U}/\mathrm{z_k}};\ \mathrm{p,k} \in (1,..\mathrm{l});\ \mathrm{p} \neq \mathrm{q} \qquad [9]$$

This information for the reference point over the efficiency frontier is enough to integrate a utility function leading to the observed decision as the optimal decision given the existing resource, economic, balance and policy constraints. For example, if we assume utility function with constant returns of scale such as the Cobb-Douglas utility function below:

$$\mathrm{U}(\tau) = \prod_{\mathrm{r}=1}^{\mathrm{l}} \mathrm{z_r}^{\alpha_\mathrm{r}}; \quad \sum_{\mathrm{r}=1}^{\mathrm{l}} \alpha_\mathrm{r} = 1 \qquad [10]$$

Then the marginal substitution relationship among any pair of attributes is:

$$-\frac{\partial \mathrm{U}/\partial \mathrm{z_p}}{\partial \mathrm{U}/\mathrm{z_k}} = -\frac{\alpha_\mathrm{p}}{\alpha_\mathrm{k}}\frac{\mathrm{z_k}}{\mathrm{z_p}} \qquad [11]$$

And the parameters of the Cobb-Douglas utility function are obtained from the following system:

$$-\frac{\alpha_p}{\alpha_k}\frac{z_k}{z_p} = \beta_{kp} \quad [12]$$

$$\sum_{r=1}^{l} \alpha_r = 1 \quad [13]$$

In Section 4 we use this type of function, which offers the advantage of having a unique solution according to the Walras' Law (a condition which is guaranteed by the constant returns of the utility function represented above). Thus the model is calibrated for each AD using the high-quality microeconomic data available in MAGRAMA (2009). This database contains data on land use, water demand, irrigation efficiency, employment (both hired and family labour), machinery and equipment, other direct costs, indirect costs, prices and yields for every crop during the period 2004-2009 and for 82% of the irrigated surface in the SRB.

3.2. Calibration errors

Farmers' decisions are simulated in accordance to the observed crop portfolio, which is the crop portfolio that maximizes the representative farmer's utility function in accordance to a set of relevant attributes. Therefore, deviations of the model's crop portfolio (x_i^*) from the observed crop portfolio (x_i^o) during the calibration stage may result in prediction errors in the model, and this is the first calibration error (e_x). The second source of error is the distance between the observed attributes and the attributes' efficiency frontier (e_f). A large distance would mean that the agent is actually taking a sub-optimal decision, and this goes against the main economic assumption that farmers are individuals that seek to maximize their utility. Finally, the third calibration error (e_τ) is the distance between the observed attributes (z_r^o) and the calibrated ones (τ_r^*). If this distance is large, it would mean that the model is not capturing the real source of utility for the representative farmer, and therefore it would be simulating someone else's utility function.

Summing up, the RPM provides three types of calibration errors that give an idea of the accuracy of the model's adjustment:

-The relative distance between the observed crop pattern and the model's one:

$$e_x = \frac{1}{n}\sum_{k=1}^{n}\left(\frac{\left(x_i^{o2} - x_i^{*2}\right)^{1/2}}{x_i^o}\right) \quad [14]$$

-The distance between the observed attributes and the attributes' efficiency frontier:

$$e_f = (\varphi - 1) \quad [15]$$

-The distance between the observed attributes and the calibrated ones:

$$e_\tau = \frac{1}{l}\sum_{r=1}^{l}\left(\frac{\left(z_r^{o\,2} - \tau_r^{*\,2}\right)^{1/2}}{z_r^0}\right) \quad [16]$$

Finally, the mean calibration error is defined as a combination of these three calibration errors:

$$e = \frac{\sqrt{e_x + e_\tau + e_f}}{3} \quad [17]$$

4. RESULTS

The methodology as above is applied to the particular case of the SRB. First, we calibrate the RPM for each one of the 12 ADs considered in the basin (the agents). Second, we conduct a simulation in which we progressively increase water prices and we assess the effects over water demand, income and employment in agriculture. Finally, we aggregate the results obtained for every AD at a river basin level.

4.1. Model calibration

Farmers have to find their optimum crop portfolio subject to a set of feasible options. It is reasonable to think that farmers will choose that crop portfolio that maximizes their income and minimizes their risk and management complexities. Accordingly, we consider the following variables in our model:

i) Expected profit per hectare, measured by the gross variable margin:

$$z_1(x) = \sum_i x_i \pi_i \quad [18]$$

Where π_i is the gross variable margin per hectare of the crop i.

ii) Avoided risk, measured by the difference between the risk associated to the crop decision $\bar{x}$ leading to the maximum expected profit ($\bar{\sigma}$) and the risk associated to the alternative crop decision x ($\sigma(\pi(x))$):

$$z_2(x) = \bar{\sigma} - \sigma(\pi(x)) \quad [19]$$

Where $\sigma(\pi(x)) = x^\tau VCV(\pi(x))x$, being $VCV(\pi(x))$ the variance and covariance matrix of the per hectare crop profits ($\pi(x)$) of the crop decision x.

iii) Total labour avoidance, the first way to measure management complexities avoidance through the reluctance to use too much labour (both hired and family labour).

$$z_3(x) = \overline{N} - N(x) \qquad [20]$$

Where $N(x) = \sum_i x_i N_i$ is the total labour used per hectare, being N_i the total labour required per hectare for a crop i, and $\overline{N}$ is the labour required to implement the crop decision leading to the maximum expected profit.

iv) Hired labor avoidance, the second way to measure management complexities avoidance through the reluctance to use too much hired labor.

$$z_4(x) = \overline{H} - H(x) \qquad [21]$$

Where similar to previous case $H(x) = \sum_i x_i H_i$ is the total hired labor used per hectare, being H_i the total hired labor required per hectare for a crop i, and $\overline{H}$ is the hired labor required to implement the crop decision leading to the maximum expected profit.

v)Direct avoided costs, the third way to measure management complexities, which includes all the seeds, fertilizers, hired equipment and all the other intermediate expenditures required to implement a particular crop decision.

$$z_5(x) = D(x) - \overline{D} \qquad [22]$$

Where $D(x) = \sum_i x_i D_i$ is the direct cost of a crop decision x, being D_i the direct cost per hectare for a crop i, and $\overline{D}$ is the direct cost required to implement the crop decision leading to the maximum expected profit.

As a result, our Cobb-Douglas Utility Function adapts the following form:

$$U(z_1, z_2, z_3, z_4, z_5) = z_1^{\alpha_1} z_2^{\alpha_2} z_3^{\alpha_3} z_4^{\alpha_4} z_5^{\alpha_5}; \quad \sum_{r=1}^{5} \alpha_r = 1 \qquad [23]$$

Where there are five unknown variables $(\alpha_r; r = 1, \ldots 5)$. Following the methodology above, we assess the relevance of each attribute by estimating the values of the alpha coefficients for every AD. These coefficients are used to calibrate the Cobb-Douglas Utility Function. Finally, we also obtain the calibration errors for every AD. The results are displayed in Table 1:

Table 1. Alpha coefficients and calibration errors

AD/Variable	α_1	α_2	α_3	α_4	α_5	e_f	e_τ	e_x	e
Sierra Segura	0.24	0.09	-	0.23	0.44	13.06%	4.64%	13.61%	7.19%
Hellín	0.13	0.52	-	0.35	-	7.34%	1.21%	7.13%	3.62%
Meridional	0.29	0.09	0.15	0.11	0.37	12.23%	5.25%	9.59%	5.47%
Vinalopó	0.38	0.03	0.20	0.07	0.32	2.42%	2.24%	2.04%	1.51%
Nordeste	0.56	0.06	0.05	0.09	0.24	8.35%	6.77%	5.29%	4.29%
Noroeste	0.18	0.11	0.30	0.41	-	8.31%	3.75%	5.30%	3.25%
Centro	0.25	0.01	-	-	0.74	6.06%	2.61%	7.36%	3.91%
Río Segura	0.99	0.01	-	-	-	7.23%	5.14%	7.41%	4.51%
Campo de Cartagena	0.39	0.16	0.34	0.08	0.02	26.79%	6.97%	21.42%	11.26%
Suroeste-Valle Guadalentí	0.36	0.30	-	-	0.33	18.02%	5.33%	16.77%	8.80%
Bajo Almanzora	0.33	0.31	0.25	-	0.11	41.16%	16.81%	41.18%	22.24%
Vélez	0.29	0.01	0.58	0.11	0.01	2.27%	11.09%	21.70%	12.19%
Average	0.36	0.14	0.16	0.12	0.22	12.77%	5.98%	13.23%	7.35%

Source: Own elaboration

There are only two attributes present in the utility function of every AD: expected profit (z_1) and avoided risk (z_2), though the former has a higher relevance in explaining farmers' decisions. The *alpha* coefficient for the expected profit has a value over 0.2 in all the ADs with the exception of Hellín (0.13) and Noroeste (0.18). This attribute is of special relevance in the Río Segura AD (0.99), where it explains most of the farmers' decisions (avoided risk has only a marginal relevance, with an *alpha* of 0.01).

On other hand, avoided risk is relevant in the Hellín AD (0.52) and in the highly productive and drought exposed ADs of Campo de Cartagena (0.16), Suroeste-Valle Guadalentí (0.30) and Bajo Almanzora (0.31).

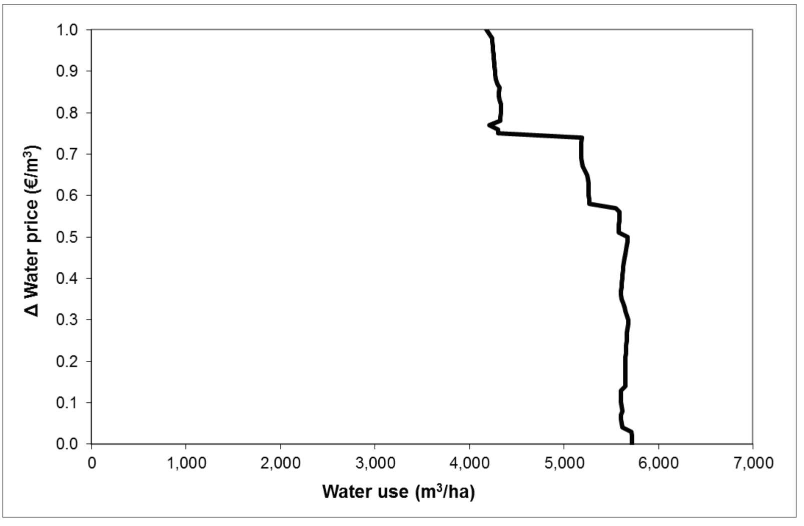
Avoided management complexities (z_3, z_4 and z_5) are relevant in a number of ADs, especially in Sierra Segura (where the sum of the *alpha coefficients* of the avoided management complexities attributes equal 0.67), Meridional (0.63), Vinalopó (0.59), Noroeste (0.71), Centro (0.74) and Vélez (0.70).

Our model shows in general low calibration errors. Most of the ADs have a mean calibration error below 10% (Sierra Segura, Meridional, Suroeste-Valle Guadalentí) and many below 5% (Hellín, Vinalopó, Nordeste, Noroeste, Centro and Río Segura). The mean calibration error is above 10% in the ADs of Campo de Cartagena (11.26%), Vélez (12.19%) and especially in Bajo Almanzora, which shows a calibration error of 22.24%.

4.2. Simulation and results

Using the utility functions above we implement a simulation in which we progressively increase conventional water prices in all the ADs of the SRB and we study farmers' responses in terms of water use, gross margin, employment generation and gross value added. We consider a price increase that ranges from 0 (baseline scenario) to 100 Eurocents/m^3 (Δ 1 €/ m^3). Results are aggregated at a river basin level[14] to obtain the water demand curve of the SRB (Figure 2). The ratio of water use over water consumption remains constant, since the high irrigation efficiency already present in the area (above 90% in many ADs) discourages further investments on the improvement of irrigation systems. Therefore, our results do not show the presence of a *hydrological paradox*.

Figure 2. Water demand curve in the SRB (m^3/ha)



Source: Own elaboration

Figure 2 represents the average water demand in m^3/ha for the whole SRB. Water demand in the basin is highly inelastic for price increases below 0.5€/m^3, and only starts showing a significant reduction in the water use for price increases above this threshold. At this point, the more water intensive crops start to be replaced by rainfed crops.

This does not mean that an increase in water prices does not have significant impacts

[14] In the case of inter-basin ADs, we applied coefficients based on the percentage of the surface of the AD that is located in the SRB.

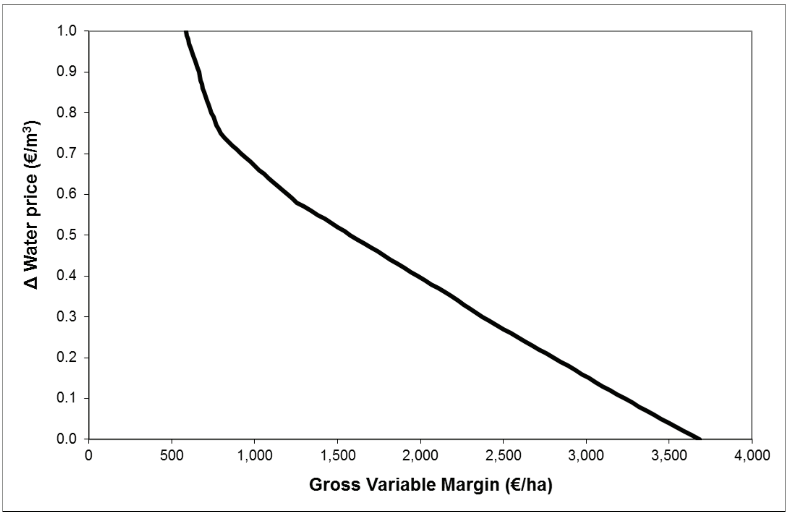
over water use below a value of 0.5 €/m^3. For example, in the upstream ADs of Hellín and Sierra Segura, a price increase above 0.13 and 0.28 €/m^3, respectively, would cause the substitution of all the irrigated crops in these areas by rainfed crops, with a much lower income (Pérez-Blanco et al., 2011). However, as a result of the low relevance of these ADs in terms of total water use (less than 100 h m^3/year), this effect is diluted when conducting an assessment at a river basin scale. Nonetheless, from an equity perspective, this impact is by no means negligible.

It is important to note that a water price over 0.6€/m^3 is unrealistic for a river basin where the average bulk water prices are below 0.1 €/m^3, since it would imply a price increase over 600%. Consequently, we should not expect a significant effect of water pricing policies over total water use in the SRB. However, an average water price increase of 0.26 €/m^3 would balance out the prices of conventional resources (0.096 €/m^3) and desalinated water (0.36 €/m^3) (SRBA, 2013), provided that the latter are kept subsidized. This could be used to promote a substitution of the overexploited conventional resources by the largely idle desalinated water, thus improving the quantitative and qualitative status of the continental water bodies in the SRB.

On the other hand, this highly inelastic water demand curve constitutes a strong argument in favour of more ambitious cost-recovery policies in the area. (Financial) cost-recovery levels in Spain have been traditionally below 100%, and in the SRB they range between 54.08% and 80.82% (Maestu and Villar, 2007). Given the large amounts invested to increase water security in the agriculture of the SRB and in the light of our results, it may be reasonable to aim towards a progressive increase of the cost recovery ratio in the basin (always considering that the asymmetric impact over the different ADs explained above need to be balanced).

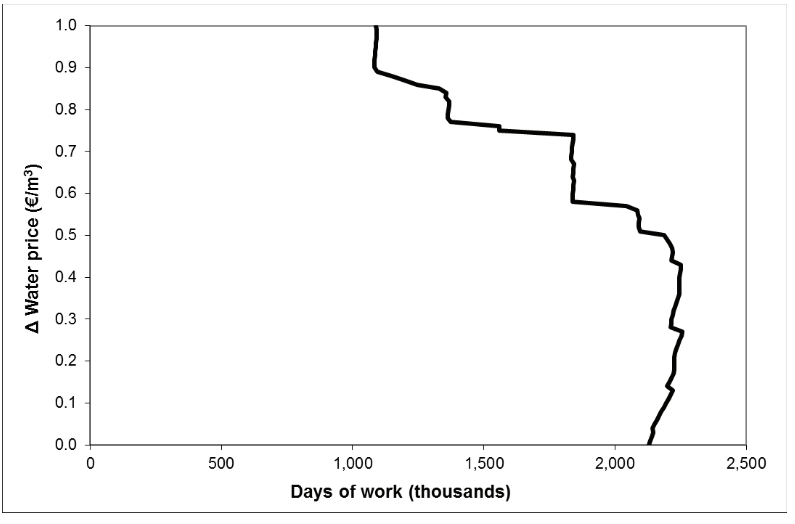
However, even if water demand is not significantly altered, a higher water price may negatively affect gross margin and employment and thus the local economy. In Figures 2, 3 and 4 we show the expected impact of a higher water price over Gross Variable Margin (GVM) (Figure 3), agricultural employment in the SRB (only hired labour, family labour excluded) (Figure 4) and Gross Value Added (GVA) (Figure 5).

Figure 3. GVM (€/ha) and water price increase (€/m^3)



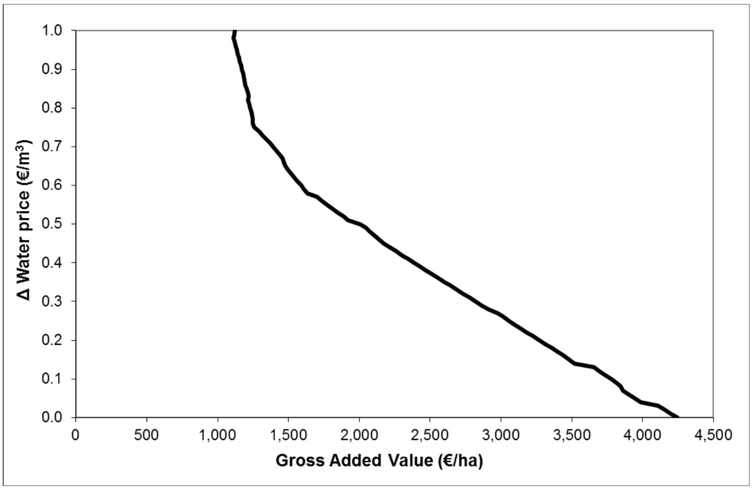
Source: Own elaboration

Figure 4. Employment generation (000s of working days) and water price increase (€/m^3)



Source: Own elaboration

Figure 5. GVA (€/ha) and water price increase (€/m³)



Source: Own elaboration

Higher water prices have initially a small positive effect over agricultural employment. Water is progressively replaced as a production factor by labour, and more labour intensive crops generate a higher agricultural employment. This happens until water prices hit 0.4 €/m^3. Rather the opposite, a water price increase has a negative effect over GVM and GVA. With a price increase below 0.4 €/m^3, farmers maintain water use and employment at levels close to those observed in the baseline, but at the expense of significantly reducing their GVM and thus also the GVA of irrigated agriculture. This means that a price increase of up to 0.4 €/m^3 mostly results into a transfer of farmers' GVM to the public sector. Above this price increase, employment, GVM and GVA fall.

5. CONCLUSION

Within the EU policy context, it is widely believed that higher (also called *right*) water prices reduce water demand (EEA, 2013; Bogaert et al., 2012; EC, 2012, 2008, 2000). Therefore, water pricing has been traditionally regarded as an effective means to reduce water use in overexploited basins such as the SRB. However, our results show that in highly profitable ADs such as those located in the SRB, farmers may react to higher prices by reducing their gross margin instead of reducing water use. In our simulations, higher prices reduce the gross margin of the farmers, who maintain water use in similar levels and may even hire more workers to compensate for marginal water use reductions.

Consequently, water pricing may become largely ineffective as an environmental policy precisely in the areas where water is scarcest. The basic lesson that can be drawn from this and similar evidence is that water prices *per se* are not *right* or *wrong*, although these adjectives have been common in EU water policy. The efficiency and effectiveness (i.e., the rightness) of pricing policies needs to be assessed based on its ability to reach the collectively agreed objectives of EU water policy at a minimum cost (EC, 2000). This depends as much on the type of economic instrument being used (water pricing or any other) as on its design and the context in place (comprising a wide array of institutional, legal and socio-economic factors).

Water policy should not confuse its goal with the instrument: aiming towards higher water prices *per se* may end up reducing farmers' income and increasing fiscal revenue without any real environmental impact (in our case, until an unrealistic price increase over 600%). From a water policy perspective, this should be deemed as a failure. On the other hand, if we design a policy mix that promotes the use of available desalinated water at the same time that prices are increased, we may end up replacing a relevant share of conventional by non-conventional water resources (desalination capacity in the SRB equals 1/6 of the average annual water demand of 1,900 million cubic meters, but it is currently used below 20%). Such a policy could include a combination of subsidies (to the users of desalinated water, for example through lower prices) and higher water prices (to the users of conventional resources), but this is case sensitive and also needs to take into account the particular institutional setup of the study site. Further research is necessary in this direction.

On the other hand, our findings support the use of water pricing as a tool to increase cost-recovery ratios and mitigate the large budgetary deficits of water authorities in Spain. Although the impacts of this policy over ADs may be asymmetric, the public sector could partially redistribute the acquired revenues through reduced taxes and higher subsidies in negatively affected areas.

Another relevant policy implication from the financial point of view concerns the allocation of agricultural subsidies in the SRB and other highly profitable basins. Subsidies to the agricultural sector in OECD countries still represent 22% of the agricultural income, and over 50% of these subsidies are considered to directly distort trade and competition (OECD, 2010). Considering the large gross margins observed and the little impact of water pricing policies over farmers' decisions, it could be advisable to review the allocation of these distorting subsidies in some areas.

The methodology developed in this paper is flexible and can be used to assess the impact of different agricultural policies over farmers' decisions, in the SRB or in other river basins where the necessary data is available. Future research should try to find a way to improve the current results through minimizing calibration errors. In our case study, calibration errors were above 20% in the Bajo Almanzora AD, and above 10% in the Campo de Cartagena and Vélez ADs. The solution to this problem may consist, for example, in finding new attributes that help explain better farmers' behaviour or in finding a more suitable utility function form.

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Part IV

Conclusions

6 General conclusions and recommendations

Developing and implementing effective water policies is troublesome. First of all, water is a unique commodity. It is an essential good with many sequential uses that intersperse public and private uses in an often complex hydrogeological system. Besides, water is a heterogeneous and finite good, and this is aggravated by its bulkiness (it is not always possible to transport adequate water from the source to the potential user at an acceptable cost). All this makes water management a complex task. In addition, water management has become increasingly complicated over time as a result of the poor performance of supply and C&C policies that have been implemented precisely to address the water crisis. Shockingly, these policies have gone on for decades, resulting in a systemic policy failure that has left several regions worldwide facing a potential environmental catastrophe.

This model, with its insistence upon increasing the system's capacity through the use of already overexploited water sources, has contributed to exacerbate the water supply crisis induced by climate change. Furthermore, it has failed to put in place the necessary incentives to drive water demand towards the collectively agreed goals of water policy. This is a complex task of paramount importance: even if water policy goals are collectively agreed by the society, experience shows that individuals with common objectives cannot be always counted on to act voluntarily to achieve them. If the appropriate incentives are missing, agents may incur in free riding (individuals who do not contribute individually and still benefit from the efforts of the others) or rent seeking behavior (individuals who benefit from collective action and throw the costs on others). This may end up threatening the sustainability of the system. Therefore, the challenge is to find suitable tools that motivate collective action through the use of incentives.

Economic instruments have the potential to provide powerful incentives for individuals to adopt certain behaviors that favor the collectively agreed goals of water policy. In spite of sharing this common ability, economic instruments are far from being homogeneous. As explained in the previous sections, there is a wide variety of economic instruments, including market (e.g., water pricing, water markets, drought insurance, subsidies) and non-market instruments (e.g., voluntary agreements). While some of these instruments are still proposals, others have been already tested in many areas worldwide, with different results. For example, in the case of water markets, one can draw a thick line between the disappointing environmental performance in Australia and Chile and the promising outcomes achieved in Spain and the US up to this point. This means that economic instruments are not a *panacea* for water management problems. Instead, they are creatures of **design**. Moreover, their final outcome also depends on the **context**, i.e., on the *policy mix* and the *institutional setup* in which they develop.

6.1 Design challenges

Economic instruments can be designed in many different ways, and therefore may attain many different results. This means that in order to fully use the potential of economic instruments to translate existing opportunities into real outcomes in terms of protecting water resources, some drawbacks in traditional policy making need to be overcome. The most important are listed below:

Cost saving does not mean revenue raising tools. Economists understood long ago that economic incentives have the potential to improve the environmental status at a cost below that imposed by traditional C&C and supply policies. The experience of different countries with economic instruments over the past decades reinforces this point of view (NCEE, 2001; Stavins, 2003; Strosser et al., 2013). Although in some cases cost reductions do not materialize to the extent expected, it is generally acknowledged that economic instruments are usually more cost effective than their alternatives. However, this just means that financial resources can be spared through the use of economic instruments, but not always raised. Focusing on the latter may end up morphing an economic instrument for water management into a financial instrument without any positive environmental impact. This confusion needs to be clearly overcome. Economic instruments for water management must be primarily addressed towards attaining the collectively agreed environmental goals in water policy; otherwise, these may not be achieved at all. This confusion regarding the priority objective of economic instruments largely explains why the vast majority of economic instruments applied in the EU so far had limited to no direct impact on water users' behavior and on the status of aquatic ecosystems (Strosser et al., 2013).

Economic instruments should be visible. In some instances it is difficult to quantify the direct environmental gains that result from the use of specific economic incentives, let alone the indirect improvements such as those in human health. This is because the performance of a given economic instrument depends on many factors that interact with each other: the very design of the economic instrument, the policies with which it is combined (i.e., the policy mix), the institutional setup and also the macroeconomic trends existent at the time it is implemented. However, there is little doubt that economic instruments are providing a new and unique element to environmental management, which in many cases results in direct and indirect benefits beyond what is possible with conventional policies. Quantifying the impact of economic instruments then becomes of paramount importance to build a sound knowledge-base for successful policy making. This can be used to anticipate the effect that economic instruments may have in areas that resemble the conditions of previous study sites. Furthermore, although *ex-post* data is a good start, some of the most promising instruments are yet to be implemented (e.g., drought insurance). This demands also the development of methodologies that allow for an *ex-ante* assessment. This thesis intended to advance in this direction, but additional effort is needed.

Transparency matters. Economic instruments may contribute to improve the technical efficiency of water use and thus offer a technical solution to disputes over competing uses of the resource (e.g., water pricing may encourage the adoption of modern irrigation technologies). Technical efficiency, though, is only part of the policy dilemma. Rebound effects and other undesirable outcomes may appear, threatening the ultimate objective of saving water. It is therefore crucial to introduce economic instruments through a meaningful dialogue with stakeholders. Acceptance of economic instruments and policy structures by water users requires transparency regarding the design of the instrument.

Dynamism is the key. Economic instruments need to be dynamic if they are to drive a change from a static to an adaptive water policy. However, this has not been the case so far. For example, although the Article 9 of the EU WFD required water pricing policies to contribute to the environmental objectives of water policy (EC, 2000), in reality European countries have focused on financial cost recovery and little has been advanced towards environmental cost recovery. Experience has shown that, once adopted, economic instruments may face rigidities (rent seeking practices, free riding behavior and other constraints) that resemble those faced by conventional policies. This is not to say that economic instruments are equally unable to attain an adaptive water policy; in fact, due to their limited dependence on infrastructure development, economic instruments have a crucial advantage over conventional policies in the avoidance of sunk costs.

Designed to minimize transaction costs. Transaction costs may block the adoption of socially desirable water policies. This is because policy makers tend to perceive transaction costs (especially the bargaining costs required to come to an acceptable agreement with all the parties involved) as being larger than environmental costs (Martin et al., 2008), thus delaying the implementation of the necessary policy reform. This perception is explained by the different barriers and obstacles to water policy reform that stem from the vested interests of some important water users (asking for financial support to overcome water management problems, instead of promoting a sustainable water use). Eventually, institutions tend to overcome these barriers driven by economic efficiency, although the transition is far from being automatic and smooth. A well designed economic instrument that minimizes transaction costs may considerably shorten this transition period and also minimize the negative environmental impacts of delaying the water policy reform.

6.2 The relevance of the context: towards an effective policy mix

If properly designed, economic instruments may be able to overcome many of the failures that are in the origin of some disappointing water policy performances. However, the effectiveness and efficiency of economic instruments to attain the water policy goals also depends on the context in which they are implemented. The particular role of an instrument cannot be understood in isolation, but as an integral part of a package (the *policy mix*) designed as an element of a major change in water policy. Besides, this whole performance is conditioned to the *institutional setup* in force.

The *institutional setup* plays a relevant role in the performance of economic instruments. Its adaptation (or alternatively, the adaptation of economic instruments to it) might be a precondition for success. For instance, the way water use rights are defined in some countries may not allow for ordinary water trading, thus increasing transaction costs through costly negotiation processes involving high-level official decisions. In the same way, drought insurance for irrigated agriculture may only be feasible in those countries where insurance systems are sufficiently developed (e.g., countries where drought insurance for rainfed agriculture already exists).

Also an adequate *policy mix* is a prerequisite for water management to succeed. Up to this point we have stressed that this policy mix should include economic instruments, but it should not be limited to them either. Economic instruments are by no means substitutes for supply and C&C policies, but tools that can strengthen water governance, i.e., complementary instruments. For example, drought insurance in Spain would be unconceivable without the existence of DMPs and River Basin Management Plans, both being C&C policies. Similarly, water markets worldwide would not be possible without water transport infrastructures, which have been developed and maintained over time through supply oriented policies.

Furthermore, although one particular instrument might seem to be better suited for a particular objective, if properly designed, each instrument can generate positive spillovers (e.g., drought insurance directly reduces agricultural water overexploitation during drought events, but it may also help to stabilize agrarian income and to regain the control over groundwater bodies on which urban users also rely). In addition, these synergies are often reciprocal (e.g., water pricing would allow better functioning water markets, while water trading would reduce the cost of water security –and thus water prices).

The basic lesson to be drawn is that rather than being *silver bullets* to solve the problems of water management, economic instruments are key components of adaptation strategies that, working under a particular institutional setup, need to be designed and implemented in combination with other policies (either economic instruments or conventional policies) so as to exploit their self-reinforcing advantages. This may significantly improve the cost-effectiveness of water policy. Noteworthy, apart from these internal synergies, a successful policy mix may also have external spillovers; these include improvements in human health, the promotion of gender equity and school attendance rates in developing countries and an improved response to the food and energy crises, among others. Relevant byproducts and ramifications stem from water policy.

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