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Robust Institutional Design Principles for Sustainable Water Markets.

Water Scarcity and Droughts (The interconnected Tagus and Segura river basins, Spain)

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1. Introduction

Water markets have been developed around the world as an instrument to deal with water scarcity problems and improve water allocation among users. Examples of countries where water markets have been studied by many authors include USA, Australia, Chile, Spain, South Africa and China (for more information see Mole and Berkoff, 2009, Grafton et al. 2010, Calatrava et al., 2012 and Zetland, 2011). There are different types of water markets which Calatrava et al. (2012) summarize as follows: agricultural, i.e. transfer of water to high-value crops, inter-basin (exceptionally allowed and controversial in Spain) and inter-sector, i.e. trade between users in different sectors such as agricultural, urban, industrial/energy, environmental. Examples of inter-sector water trading schemes include water trading from agricultural to urban in South-Western USA, from agricultural to government with the purchase of water from farmers to comply with environmental standards in Australia and recently in Spain, from agricultural to energy in South Spain. Other forms of water markers include groundwater, i.e. trading of abstraction rights instead of water itself (e.g. the establishment of national groundwater framework in Australia), and water supply options contracts barely implemented in practice, i.e. one user accepts to reserve a share of his quota and to sell it to other user who might need it if certain conditions hold, for instance in drought periods.

As water markets in different countries function under different rules and mechanisms, it is difficult to conclude about their advantages and disadvantages. However, WWF (2007) (cited in Rey et al (2012)) attempts to summarise the main advantages of water markets in ensuring the allocation of water to the highest value use, providing incentives to users for water preservation and making possible additional water available for the environment without reducing overall economic activity. On the other hand, water markets are faced with some difficulties, substantially varying in transactional costs and access to information due to discrepancies in income levels and access to capital. Other difficulties which can affect the functioning of water markets include third party
effects and the fact that water markets for water rights are often not as active as spot markets.

There were several studies in the past where the costs and benefits from water trading schemes were evaluated. For instance, in Australia allocation markets have been used to manage uncertainty and risk within and between seasons, while entitlement markets have been used to adjust irrigators’ risk position in the long term, resulting in subsequent use of the allocation market to manage this new risk position (Bjornlund, 2006). Furthermore, Brooks and Harris (2008) stated that water trading in Victoria’s Watermove program generates substantial economic benefits, and the gains achieved might provide guidance on markets mechanisms for other countries. Efficiency gains in Watermove reflect the reallocation of water from low to high valued uses, promoting structural adjustment in the agricultural sector as inefficient farmers exit the sector. More recently, Zetland (2011) has provided an overview of the water markets for water quantity and quality in Europe, which are underdeveloped due to difficulties in their implementation driven by institutional constraints and high transaction costs. Moreover, Adler (2009) underlined the importance of global climate change and pressure on water resources and availability in the development of water markets. The author stated that a gradual change toward water marketing and market pricing will improve the management of water supplies, guarantee more efficient allocation of available supplies, and encourage cost effective conservation measures, thereby mitigating the impact of climate change on supplies and availability. However, Matthews (2004) argues that water markets do not function efficiently as property rights were not designed for market transactions. The author raises several issues regarding the structure of a water right system driven by the experience in the western United States. Several recommendations for understanding the structure of property rights include the separation of water rights from land, registration of water rights for a certain period and well-specified rules for
transferable water rights. Another study by Grafton et al. (2010) employed an integrated framework to assess and compare the institutional foundations, economic efficiency and environmental sustainability of water markets in Australia, the western US, Chile, South Africa and China and suggested that effective institutional arrangements and allocative mechanisms are of great importance for a well-functioning water market.

This paper discusses a framework for analysing robust institutions for water markets drawn on the new institutional economics school of thoughts which is based on Williamson, North, Coase and Ostrom theories on transaction cost economics, property rights and collective actions. Based on these theories, we review the evolution and development of water reforms and markets in countries such as in Australia (Murray-Darling), USA (California and Colorado), Chile, and in Spain (Tagus). The reason for choosing these overseas countries is that there is considerable empirical evidence and research on the evolution of water markers, which allows us to identify costs and benefits, advantages and disadvantages of introducing and development of water markets. The reason for choosing Spain as our country of study from Europe is attributed to the fact that the absence of robust water governance and effective surveillance does not allow water markets to be efficient and socially accepted (Garrido et al. 2012). Based on the lessons learned from the Spanish and international review where different legal systems exist, a list of robust recommendations for the improvement of water markets in Spain is provided followed by a discussion of the development of a robust water governance model based on Sharma’s approach (2012). Even limited in scope, the review will provide an important insight on how water markets had been evolved so far and how they could be further developed to be environmental, social and economic accepted.

This paper unfolds as follows. Section 2 provides a brief definition of the new institutional economics approach followed by a discussion of the design of robust principles for governing sustainable resources. Section 3 provides an example of these approaches in the water resource management of the Murray-Darling basin in Australia, USA (Colorado and California), Chile and Spain (Tagus river basin). Section 4 discusses a
list of robust recommendations for improving water markets in Spain followed by the introduction of an effective water governance model for this country. The final section concludes.

2. The New Institutional Economics Approach

2.1 Theory

The New Institutional Economics (NIE) approach builds of two schools of thoughts: the neo-classical economics and the institutional analysis. Under NIE, some of the unrealistic assumptions of neo-classical economics (such as perfect information, zero transaction costs, full rationality) are relaxed, but the assumption of self-seeking individuals attempting to maximize an objective function subject to constraints still holds (Sharma, 2012, Libecap, 2006, Kherallah and Kirsten, 2001). The additional constraint that NIE assumes is that institutions matter for economic performance. The institutional analysis refers to a set of formal (e.g. laws, contracts, political systems, organizations, markets, etc) and informal rules of conduct (e.g. traditions, norms, customs, sociological trends etc) that facilitate coordination or govern relationships between individuals (Kherallah and Kirsten, 2001). Therefore, the NIE (Figure 1) suggests that economic activities are embedded in a framework of informal and formal institutions, and its purpose is to explain the determinants of institutions and their evolution over time, and to evaluate their impact on economic performance, efficiency and distributions (Nabli and Nugent, 1989).
Figure 1. The New Institutional Economics Approach

Being a multi-disciplinary approach, NIE has several branches (Figure 2). These expand from new economic history and public choice & political economy (macro-level analysis) to transaction economics, theory of collective action and law and economics (micro-analysis) (for a more comprehensive definition of the NIE branches see Kherallah and Kirsten, 2001). Three components are of great significance in the NIE approach and are the focus of this study; the transaction costs economics and property rights, and collective actions. Transaction costs are defined as the costs of screening and selecting a buyer or seller, the costs of negotiating, monitoring or enforcing a contract (Coase, 1937) and, if ignored can reduce the efficiency of economic activities. A major effect of good institutions is therefore to reduce transaction costs. According to North (1997), the major challenge is to evolve institutions in which firstly, the transaction costs are minimized and secondly, the incentives favour cooperative solution, in which cumulative experiences and collective learning are best utilized (Gandhi and Crase, 2009). In the same line of thinking with Coase and North, Williamson suggested that a trade-off has to be made between the costs of coordination and hierarchy within an organization, and the costs of transacting and forming contracts in the market. This trade-off will depend on the magnitude of transaction costs (Kherallah and Kirsten, 2001).

With transaction costs, property rights play an important role for efficiency. According to J.R. Commons (1957) property rights define relationships among people regarding things (Schlager, 2005). Schlager and Ostrom (1992) defined five types of
property rights; the rights of access, i.e. to enter a defined physical property, the rights of withdrawal, i.e. to obtain the products of a resource e.g. appropriate water, the rights of management, i.e. to regulate how to use and improve the resource, the rights of exclusion, i.e. to determine who enters the resource, and the rights of transfer, i.e. to sell, lease, or leave the resource (Schlager, 2005). If property rights are not well defined then transaction costs can be high (e.g. costs of obtaining information for a good or service, bargaining and monitoring a contract). As a result, good institutions need to minimize the transaction costs of renegotiations so that a new level of efficient equilibrium of resource-use can be achieved (Coase, 1960).

The importance of property rights in relation to specific goals in water management issues was underlined by Bruns et al. (2005). After reviewing the water property rights reform in six countries, they concluded that the property rights as tools for more equitable, sustainable, and efficient water management requires better sequencing of reforms, redesigning institutions for participatory water governance, resolving tenure rights, and developing equitable arrangements for regulating transfers.

Moreover, the NIE approach takes into account the theory of collective action mainly driven by Ostrom’s work (1990). Ostrom (1990, 1994) underlined that the institutions and institutional structures developed by individuals, groups and governments to organize human activities influence the outcome of managing “common pool resources” (CPRs) (Biswas and Venkatachalam, 2010). Furthermore, new institutional economic theories suggest that institutions contributing to sustainable management of CPRs are generally efficient in nature because only the efficient ones can survive by way of crowding out all the inefficient ones (see Alchian and Demsetz, 1972) irrespective of the social outcomes (Biswas and Venkatachalam, 2010).

Therefore, by relaxing some of the assumptions such as unbounded rationality and information availability and maintaining others like the concept of efficiency i.e. minimization of transaction costs, the new institutional economics approach can deal with
a large range of phenomena, such as water resource management, including economic, political and social considerations (Sharma, 2012). The next section discusses in more detail the theories of transaction cost economics and collective action, the 4 level institutional framework by Williamson (2000) and the robustness of self-organized common-property institutions by Ostrom (1990).

2.2 Levels of institutions (Williamson)

The NIE approach distinguishes between informal and formal institutional environment and between institutional environment and institutions of governance. This is evident in Figure 3, where the 4 level of institutions by Williamson (2000) is depicted.
The solid arrows that connect a higher with a lower level indicate that the higher level imposes constraints on the level immediately below, whereas the reverse arrows that connect lower with higher levels are dashed and signal feedback (Williamson, 2000). The top level (Level 1) is the informal institutional environment which includes the customs, traditions and norms, which change very slowly. The next level (Level 2) is the formal institutional environment, which includes the constitution, the legal system, judiciary, polity, and property and contract rights. Level 2 introduces the “formal rules” of the game and opens up the opportunity for first-order economizing: get the formal rules of the game right (Williamson, 2000). “The play of the game” (Level 3) is the economic organization of contracts and governance structures; market, quasi-market, and hierarchical modes of contracting, more generally of managing transaction costs and seeing economic activity through to completion (Patibandla, 2012). Level 3 opens up the opportunity for second-order economizing: get the governance structures right (Williamson, 2000). The fourth level is the level at which neo-classical analysis works e.g. evolution of resource allocation and employment and changes continuously. The 4-level institutional analysis by Williamson can be used as a framework to evaluate the performance of institutions for water reform related aspects and moreover, to allow for the establishment of new and better “pathways for reform” of water resource management policies in the face of looming water related problems (Sharma, 2012). Section 3 discusses the application of this framework to the Australian experience, the Murray-Darling basin where several water reforms occurred with respect to the definition of water rights, development of water markets and allocations, and to cope with water over-allocation problems. Recommendations for further improving the water trading mechanism in the Murray-Darling basin are briefly discussed as well.
2.3 Robust design for self-organized common-property institutions (Ostrom)

Ostrom (1990) in *Governing the Commons* illustrated eight key design principles related to long-term robustness of institutions created to govern common-pool resource systems. Examples of common-pool resources include both natural and human-made systems including: groundwater basins, irrigation systems, forests, grazing lands, mainframe computers, government and corporate treasuries (Ostrom, 2001). Examples of the resource units derived from common-pool resources include water, timber, fodder, computer-processing units, information bits, and budget allocations (Blomquist & Ostrom, 2000).

Figure 3. Four-Levels of institutions: Source: Williamson (2000) and Sharma (2012).

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1 Common-pool resources produce finite quantities of resource units and one person's use subtracts from the quantity of resource units available to others (Ostrom, 1994, 2001).
1985 and Ostrom, 2001). The analytical framework to long-term robustness of institutions for governing sustainable resources developed by Ostrom (1990) is illustrated in Table 1.

1. Clearly Defined Boundaries
   The boundaries of the resource system (e.g., irrigation system or fishery) and the individuals or households with rights to harvest resource units are clearly defined.

2. Proportional Equivalence between Benefits and Costs
   Rules specifying the amount of resource products that a user is allocated are related to local conditions and to rules requiring labor, material, and/or money inputs.

3. Collective-Choice Arrangements
   Most individuals affected by harvesting and protection rules are included in the group who can modify these rules.

4. Monitoring
   Monitors, who actively audit biophysical conditions and user behavior, are at least partially accountable to the users and/or are the users themselves.

5. Graduated Sanctions
   Users who violate rules-in-use are likely to receive graduated sanctions (depending on the seriousness and context of the offense) from other users, from officials accountable to these users, or from both.

6. Conflict-Resolution Mechanisms
   Users and their officials have rapid access to low-cost, local arenas to resolve conflict among users or between users and officials.

7. Minimal Recognition of Rights to Organize
   The rights of users to devise their own institutions are not challenged by external governmental authorities, and users have long-term tenure rights to the resource.

*For resources that are parts of larger systems:*

8. Nested Enterprises
   Appropriation, provision, monitoring, enforcement, conflict resolution, and governance activities are organized in multiple layers of nested enterprises.

*Table 1. Design principles for governing sustainable resources derived from long-enduring studies of institutions. Source: Ostrom (1990, 2009).*
The first design principle requires that the boundaries of the resource systems and the property rights of individuals are clearly defined. This principle ensures that participants know who is in or out of a defined set of relationships and therefore with whom to co-operate (Ostrom, 2009). The second principle refers to the rules that need to be well specified such that benefits and costs are allocated proportionally to the participants. If some users pay low costs but obtain high benefits over time, then the willingness by others to participate and follow the rules reduces. The third principle denotes that the users can participate in the process of making and modifying the rules, whereas the fourth principle is about the enforcement of rules and monitoring of the resource condition by the government or locally by the self-organized resource regimes. The fifth principle refers to the penalties that must be imposed when a user violates the rules and obtains benefits in the burden of others. Ostrom (2005, 2009) states that the first five principles work together. For instance, when the participants of a resource system make their own rules (collective action agreements) that are imposed and monitored by local users (monitoring) employing punishments for breaking up the rules (graduated sanctions) that clearly define who has rights to abstract from a well-defined resource (clearly defined boundaries) and that effectively assign costs proportionate to benefits (proportional equivalence between benefits and costs), then collective action and monitoring problems can be solved in a reinforcing manner (Ostrom, 2005 and 2009). Moreover, the sixth principle states that systems with low-cost conflict resolution mechanisms are more likely to survive, whereas the seventh principle suggests that external government agencies do not challenge the right of local users to create their own institutions (Cox et al., 2010). The last principle for robust systems postulates that the rules for instance, to allocate water among larger parts of a resource system may differ from those established for small or single parts. Therefore, among long-enduring self-governed regimes, smaller-scale organizations can be nested in ever-larger organizations (Ostrom, 2009).
The eight general principles for robust systems were reviewed and updated by Cox et al (2010) based on the results from an analysis of almost 100 studies which applied Ostrom’s principles for managing common-pool resources. The improvements are related to the principles 1, 2 and 4. Therefore, the design principle 1 is separated into two parts. The first one is on user boundaries where clear and locally understood boundaries between legitimate users and nonusers are present, and the second part is on resource boundaries where clear boundaries that separate a specific common-pool resource from a larger social-ecological system are present. The design principle 2 is also split into two parts, congruence with local conditions and appropriation and provision. The former states that appropriation and provision rules are congruent with local and social environmental conditions, whereas the latter suggests that appropriation rules are congruent with provision rules; the distribution of costs is proportional to the distribution of benefits. Finally, the design principle 4 distinguishes between monitoring users and the resource. The former refers to individuals who monitor the appropriation and provision levels of users, whereas the latter refers to individuals who monitor the condition of the resource. Ostrom (2010) suggested that the improvements in the design principles 1, 2 and 4 together with the other principles are the robust principles that can ensure the probability of long term survival of an institution developed by the users of a resource.

Moreover, Ostrom’s further work focused on designing principles to deal with economic and environmental challenges that could result in the sustainability of common pool resources. She especially highlighted the problems with “the tragedy of commons” (see Hardin, 1968) for more details), which occurs as a perceived lack of incentive to keep the quality and quantity of the commons with a view towards long term sustainable usage (which necessary entails an acknowledgment that the resource is finite in the short term and only infinite in the long term if measures are taken to ensure that the resource can renew itself) (Sharma, 2012). The lack of incentives can lead to over-use and eventually
deterioration of the quantity and quality of the common resource. Therefore, Ostrom (1997 and 2001) provided an analytical framework that describes the conditions under which self-governing/localized government institution can form and manage successfully a common pool resource. These conditions which are displayed in Table 2 are separated into attributes of the resource and of the appropriators, i.e. users that withdraw resource units like water from a common pool resource.

Attributes of the Resource:

R1. Feasible improvement: Resource conditions are not at a point of deterioration such that it is useless to organize or so underutilized that little advantage results from organizing.

R2. Indicators: Reliable and valid indicators of the condition of the resource system are frequently available at a relatively low cost.

R3. Predictability: The flow of resource units is relatively predictable.

R4. Spatial extern: The resource system is sufficiently small, given the transportation and communication technology in use that appropriators can develop accurate knowledge of external boundaries and internal microenvironments.

Attributes of the Appropriators:

A1. Salience: Appropriators are dependent on the resource system for a major portion of their livelihood.

A2. Common understanding: Appropriators have a shared image of how the resource system operates (attributes R1, 2, 3, and 4 above) and how their actions affect each other and the resource system.

A3. Low Discount rate: Appropriators use a sufficiently low discount rate in relation to future benefits to be achieved from the resource.

A4. Trust and Reciprocity: Appropriators trust one another to keep promises and relate to one another with reciprocity.

A5. Autonomy: Appropriators are able to determine access and harvesting rules without external authorities countermanding them.

A6. Prior organizational experience and local leadership: Appropriators have learned at least minimal skills of organization and leadership through participation in other local associations or learning about ways that neighboring groups have organized.

Table 2. Attributes of the resource and appropriators for self-governing of common pool resources. Source: Ostrom (1997 and 2001)
Ostrom (1997) concluded that robust, long-living self-governing systems could work if the resource system is sufficiently small and appropriators can develop precise knowledge of external boundaries and internal microenvironments (R4) and the flow of resource units is relatively predictable (R3). With respect to the attributes of the appropriators, important components are a common understanding among the appropriators, which includes knowledge about the operation of the resource system and the effect of each other actions (A2) and the establishment of trust and reciprocity among appropriators (A4). The analytical framework depicted in Table 2 was further developed by Ostrom (2007) to include the ecological aspect of governing a common pool resource, therefore called social-ecological systems (SES). Ostrom provided a multi-tier framework with seven variables (see Table 3) to analyze how attributes of a resource system (e.g. lake, river), the resource units generated by that system (e.g. water), the users of that system, and the governance system, jointly impact the interactions and outcomes obtained at a particular time and place and how these may influence and be influenced by larger or smaller socioeconomic and political settings in which they are embedded as well as by a larger or smaller social-ecological systems.

<table>
<thead>
<tr>
<th>Social, Economic, and Political Settings (S)</th>
<th>Governance System (GS)</th>
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<tr>
<td>S1- Economic development. S2- Demographic trends. S3- Political stability.</td>
<td>GS1- Government organizations</td>
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<td>S4- Government settlement policies. S5- Market availability.</td>
<td>GS2- Non-government organizations</td>
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<td>Resource System (RS)</td>
<td>GS3- Network structure</td>
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<td>RS1- Sector (e.g., water, forests, pasture, fish)</td>
<td>GS4- Property-rights systems</td>
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<td>RS2- Clarity of system boundaries</td>
<td>GS5- Operational rules</td>
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<td>RS3- Size of resource system</td>
<td>GS6- Collective-choice rules</td>
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<td>RS4- Human-constructed facilities</td>
<td>GS7- Constitutional rules</td>
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<td>RS5- Productivity of system</td>
<td>GS8- Monitoring &amp; sanctioning processes</td>
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<tr>
<td>RS6- Equilibrium properties</td>
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Resource Units (RU) Users (U)
RU1- Resource unit mobility U1- Number of users
RU2- Growth or replacement rate U2- Socioeconomic attributes
RU3- Interaction among resource units of users
RU4- Economic value U3- History of use
RU5- Size U4- Location
RU6- Distinctive markings U5- Leadership/entrepreneurship
RU7- Spatial & temporal distribution U6- Norms/social capital

Interactions (I)?
I1- Harvesting levels of diverse users Outcomes (O)
I2- Information sharing among users O1- Social performance
I3- Deliberation processes measures (e.g., efficiency, equity,
I4- Conflicts among users accountability)
I5- Investment activities O2- Ecological performance
I6- Lobbying activities measures (e.g., overharvested,
Related Ecosystems (ECO) resilience, diversity)
ECO1- Climate patterns. O3- Externalities to other
ECO2- Pollution patterns. SESs
ECO3- Flows into and out of focal SES.

Table 3. Muti-tier variables in framework for analyzing a social-ecological system (SES).


3. International Experiences from Water Markets

3.1 Water Markets in Australia

The theories described in the previous sections have recently been applied in the Murray-Darling (M-D) river basin in Australia by Sharma (2012) where the performance of water reforms and institutions is evaluated over time. The author also provided further
recommendations on how to improve the water trading in this basin, and eventually, a new robust water governance model, which can fulfil economic, political, social and environmental objectives.

According to the four level analysis of institutions by Williamson (2000), the water reforms in Australia till ’90s, were focused more on level 4, i.e. on the development of water resource allocations (using the markets as a mean for allocating water). Less attention therefore was paid to make water reforms to address the levels 1, 2 and 3, i.e. to define the informal and formal rules and play of the game, for instance, water right systems and water allocation procedures or even to change social attitudes towards water, for instance, by focusing on making improvements in water quality and environment. The situation, however, changed after 1990, with the reforms induced by the Council of the Australian Governments (COAG) in 1994 which were brought within the National Competition Policy, the Murray Darling “cap” to limit water diversions and eventually to National Water Initiative (NWI) and National Plan for Water Security (NPWS) in 2004. For the purposes of this study, we are focusing only on these reforms because we believe that they can be considered as robust for the sustainability of the water resources and the well-function of the water markets.

In 1994, the COAG agreement included several important recommendations. Increasing emphasis was placed on the development of water markets and improving water allocation arrangements (especially, for environmental usage) (Sharma, 2012, McCay, 2005). The former included recommendations for separation of water licenses from land title, allowing water access entitlements and allocations to be deployed to uses generating greater economic returns (COAG, 1994, Young and McCoil, 2002 and 2003). This was the first step to robust institutional arrangements for water allocation and management and as Young and McCoil (2002, 2003) suggested, the clue to the robust resolution of many of Australia’s water resource problems lied more with separation than
in integration. With respect to improving water allocation arrangements for environmental usage, it was suggested that environmental requirements should be taken into account and the environment should be treated as a legitimate user of water. Both changes recommended by the COAG agreement and further adopted by the National Competition Policy can be considered as changes on level 2 and 3 institutions (Williamson), as the “formal” rules (Institutional Environment) and “play of the game” (Governance) were clearly and well-defined. They can also be defined as robust in accordance with Ostrom’s design principles 1 and 2 (or equivalently, the updated list suggested by Cox et al. 2010 and further approved by Ostrom as well). Therefore, unbundling commenced in 2000 in the State of New South Wales and was formally required under the National Water Initiative established in 2004. The term “property right”, was redefined and described as “water access entitlement” as it was easier to talk about the nature of each person’s entitlement and avoid getting tangled up in debates about the nature of people’s rights (Young, 2011). Young (2011) summarizes the robust unbundling water allocation regime as follows. Access entitlements took the form of a share and were usually issued in perpetuity. Once the system was set up the only way to secure an entitlement to a share of water in a system was to purchase a share from an existing shareholder. Ownership of entitlements was vested in individuals and arrangement put in place to enable water to be traded from one irrigation district to another. Allocation trades were implemented by debiting one person’s water account and crediting another person’s water account. Entitlement trades were implemented by amending names on a water entitlement register. Entitlements could be mortgaged and

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2 Young and McCoil (2002, 2003) suggested that robust institutional arrangements for water management and allocation could be achieved based on Tinbergen principle (1950), which states that to attain a given number of independent policy targets through time there must be, at least, an equal number of policy instruments. Therefore, the components of the existing allocation licensing regime would be robust if they are separable from each other. Moreover, based on Tinbergen principle, Young (2012) in a recent report suggests a new robust allocation licensing mechanism for the England and Wales, where an unbundling regime would include a long-term tradeable entitlement issued in perpetuity, a short-term tradeable allocation and a non-tradeable use, abstraction and discharge permit.
finally, brokers were employed to bring buyers and sellers together and dealt with each trade.

In addition to this, the introduction of a ceiling, called “cap” in 1994, for diversions from the Murray-Darling river system aimed to protect and enhance the riverine environment and eventually, to meet ecological and social needs. This reform placed the environment at the centre of policy making thought process (level 1 and 2 institutions) and affected the governance rules (level 3 institutions) for water allocation (Sharma, 2012). It can also be defined as robust according to Ostrom’s design principle 4 (monitoring or equivalently, monitoring the resource). Moreover, the National Water Initiative established in 2004 although focusing on water markets and trading, can be considered as the first initiative that attempts to cover all 4 levels of institutions and mostly 2, 3 and 4 by including key policy areas such as best practice water pricing and institutional arrangements, water resource accounting, water access entitlements and planning framework and community partnerships and adjustment (National Water Commission, 2007b). Although the NWI can be denoted as robust based on Ostrom’s design principle 1, it did not address other important components such as mechanisms to deal with conflicts between unaffected parties (Ostrom’s design principle 6) or to improve the participation in the water markets (Sharma, 2012). Finally, the National Plan for Water Security (NPWS) addressed the over-allocation problems which were created by the development of water markets with a purpose of guaranteeing that environmental assets received an appropriate allocation of water for regeneration. This reform, where the government buybacks entitlements for water allocation for the environment, can be considered as a change to level 1 and 2 institutions since the environment is the centre of policy making process and robust in accordance with Ostrom’s principle 4 (monitoring or equivalently, monitoring the resource).

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3 Over-allocation occurs when not enough water is allocated for environmental “regeneration”.

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Moreover, Sharrma (2012) further applied Ostrom’s analytical framework with respect to the attributes of the resources and appropriators (see at Table 2) in the Murray-Darling river basin. The author stated that the M-D basin is sufficiently large and the flow of resource units is not predictable and there is no common understanding and trust among the appropriators. In contrast, there is room for feasible improvement in the resource system, reliable and valid indicators of the condition of the resource system are frequently available at a relatively low cost. Moreover, the appropriators are dependent on the resource system for a major portion of their livelihood, they use a sufficiently low discount rate in relation to future benefits to be achieved from the resource and have learnt at least some minimal skills of organization through participating in other local associations.

Overall, it is concluded that it is of great importance an increased level of priority for establishing the informal and formal rules of the game and the play of the game to satisfy robust design principles for sustainable use of resources and then the markets can be used as a mean to allocate water. Sharma (2012) moved a step forward by providing some policy recommendations with respect to the improvement of water markets in the Murray-Darling river basin. The first proposal refers to the establishment of market entry restrictions in the temporary market trade only and not to the permanent trade in order to avoid market distortions created by the “activation of sleeper entitlements”\(^4\). The advantage of the first proposal lies on the fact that there could be a better specification of the nature and number of participants in the temporary trade. The second proposal refers to the reduction in institutional transaction costs such as the time taken to process different types of water trading (permanent and temporary). For instance, currently the approval time for water access allocation varies across the basin, from 7 days in Queensland and 30 working days in New South Wales and South Australia. The final

\(^4\) Sleeper entitlements are those entitlements that were not previously used. Once the water trading occurs, it is likely that the users of those entitlements would be willing to take part in the market to obtain some financial gain (Sharma, 2012).
suggestion is related to the reduction of monetary transaction costs, e.g. administrative costs, water use monitoring and enforcement costs by introducing uniformity of fees and charges across different jurisdictions. This proposal could incentivize participants between different states to take part in trade. For instance, as it stands now, in the New South Wales the application fee for approval for temporary and permanent trade is $75 and $250 respectively, whereas the fee for registration of trade is $73.25 for the permanent trade. Moreover, in South Australia, the application fee for approval for temporary and permanent trade is even higher, $205 and $349 respectively, whereas in Victoria the application fee for approval for temporary and permanent trade is $70 and $150 respectively, and the fee for registration of trade is $101 in the case of a permanent trade.

With respect to dealing with conflicts between disaffected parties in the M-D basin, it has been proposed a shift towards more centralism approach. This implies that the Federal government should be responsible for the rules and the play of the game, with consultation from the state level representatives, whereas the responsibility of water resource allocation would reside with the state governments (Sharma, 2012).

Although there were several robust institutional arrangements in the M-D basin for water allocation and management, water over-allocation problems occurred and now the environment is on the centre of attention for the policy makers. Therefore, a new robust governance model for the Murray-Darling river basin must be introduced. According to Rogers (2002), governance refers to “the capability of a social system to mobilize energies, in a coherent manner, for the sustainable development of water resources and includes: i) the ability to design public policies (and to mobilize social resources for their support) which are socially accepted, whose goal is the sustainable development and use of water resources; and: ii) to make their implementation effective by the different actors/stakeholders involved in the process” (Solanes, 2012). In the case of the M-D river basin, the new robust governance model – based on the theories of Williamson, North and
Coase on institutions and transaction cost economics, and Ostrom’s work on robust institutions (Tables 1 and 3)- should give the environment the highest priority in the policy making process and implementation and second priority should be given to social and economic needs which then can denote the role the political components needs to play in facilitating the realization of the environmental, social and economic objectives (Sharma, 2012).

3.2 Water markets in California and Colorado (USA)

This section discusses the water markets and legal change in California based on the study by Brewer et al. (2008) where the New Institutional Economics approach is implemented. This section also includes a discussion of the development of water markets in Colorado where inter-sectoral transfers had positive impacts from an environmental, economic, and social perspective. The empirical evidence is based on the studies by Howe and Goemans (2003) and Howe (2011) and the water reforms based were evaluated based on the framework analysed in the previous sections.

Following the New Institutional Economics (NIE) approach, Brewer et al. (2008) looked into the interactions among regulation, property rights, and water markets in California over the period 1987–2005. The authors examined whether and how the definition of water rights and the regulation of water transfers have affected observed market activity in the extent and pattern of water trades and their duration, and the nature of the contracts used (short-term leases, long-term leases, and sales). Over the period of study, the authors identified the changes in law that either strengthened or weakened property rights to water and raised or lowered the transaction costs of trading. It was mentioning that the greatest activity for legal change to support water markets were in the years 1987, 1988, 1991, 1993, 1999, and 2003, whereas changes that limited water markets occurred in 1988, 1999, and 2001 (for more detail see Brewer et al., 2008). For instance, in 1991 the Drought Water Bank of California was created to facilitate temporary transfers of water from agricultural to the urban sector, at a single price set by the state government,
through different types of contracts (Level 3 institutions) (Rey et al, 2012). The results indicated that most active and subsequently robust factors in support of markets were making the water right more precise such as defining beneficial use to include trading activities, allowing for the transfer of water rights, separating water from the land for trading, and defining conservation and the trading of conserved water. In contrast, the most active factors limiting water markets were restrictions on transfers to protect other water users, restrictions to protect the environment, requiring third-party compensation, requiring notice of transfers, and allowing for third-parties to protest and challenge proposed transfers (Brewer et al., 2008).

Moreover, in the western USA, the water rights are defined as appropriation rights meaning that they are obtained on the basis of beneficial use rather than land ownership (Hodgson, 2006). Under the appropriation system of water rights, all water withdrawn from the natural setting is represented by a “water right” or by a groundwater pumping permit, owned by individuals, municipalities or water companies, and the shares of these organizations can be therefore bought or sold (Howe and Goemans, 2003). This type of rights is in contrast to the riparian rights which refer to the situation where both land and water need to be purchased if water is used for another purpose. Below we will focus on the characteristics of the water rights, and the benefits from intersectoral trading between the two regions of the Colorado basin, the South Platte and Arkansa, where water is transferred through the federal Colorado-Big Thompson (C-BT) project.

As part of this project, the Northern Colorado Water Conservation District (NCWCD) was founded whose responsibilities were on the diversion works of the project and the allocation of water on the eastern side of the mountains (Howe, 2011). The U.S. government continued to be the owner of the water and of the water, but the District owned the right to use allocate all the water made available by the C-BT project as long as it meets repayment obligations and operates and maintains the project facilities as
stipulated in the repayment contract (World Bank, 1999). NCWCD shares have unique characteristics that make water trading activities very attractive. Firstly, they are homogeneous units (each share gets the same amount of water and there are no priorities) (Level 2 institutions). For instance, the amounts of transfers were annually collected and were classified by size and nature of seller and buyer (e.g. agricultural to urban or agricultural to agricultural). Secondly, the water district holds the rights to all return-flows and thirdly, transfers do not have to pass through the water court but require only the approval of the NCWCD board (Level 3 institutions) (Howe and Goemans, 2003, Molle and Berkoff, 2009). Therefore, in this case well-defined property rights strengthened the development of water resource allocations and kept the transactional costs low resulting in significant economic and environmental benefits for the participants.

The results indicated that the economic impact, both directly and indirectly, per acre foot of water transferred from agricultural to urban uses within the basins, was very positive. Moreover, although there was an increasing rate of share of ownership over time for the cities and industry, the share in actual use is not increasing rapidly as cities “rent” some of their water back to agriculture on an annual basis, subject to recall in drought years (Howe and Goemans, 2003). The volume and direction of rentals depend on weather conditions, for instance, during drought period cities may withhold water from agriculture and charge higher prices (Howe, 2011). Also, the NCWD preserved the water resources by paying attention to the preservation of productivity of agriculture lands, water quality in soil, aquifers and maintenance of ecosystem services. Hence, within the Colorado-Big-Thompson system, robust institutional arrangements such as the homogeneous nature of water shares and the avoidance of water court review approval for water transfers, allowed buyers and sellers to carry out small transactions as the need arises rather than occasional large transfers. It is therefore concluded that the efficient and continuous water market within the Colorado-Big-Thompson system fulfils Ostrom’s design principles 1, 2, 4 and 6. New institutions, i.e. well-defined property rights, were successfully adapted to the specific circumstances and needs of the region (local level),
homogeneity of water shares allowed water transfers among groups with the same needs and concerns and protection of the open-access resources and finally, any water transfer approval was dealt locally (Ostrom, 1990, Ostrom et al. 1993, and Ostrom and Gardner, 1993). However, considerable attention needs to be paid to the case when there is an out of basin water transfer. If the region where the water is taken from is economically depressed, then a water transfer might cause difficulties in its financial vitality (e.g. loss in agriculture production, income, employment). Therefore, a transfer fee per acre foot could be imposed on the buyer and transferred to a unit of general government in the area of origin to support social services during the period of transition (Howe and Goemans, 2003 and Howe, 2011).

3.3 Water markets in Chile

This section discusses the development of water markets in Chile based on the study by Donoso (2011) evaluating the water reforms based on the new institutional economics approach analyzed in the previous sections. In Chile, the government introduced neo-liberal economic policies which supported private property rights and free markets through the establishment of the National Water Code (WC). The 1981 WC maintained water as “national property for public use”, the water rights were separated from land and granted transferable water-use rights (wur) to individuals by the Directorate General of Water (Dirección General de Aguas, DGA). These water use rights allow a person to have a certain water flow of a river or aquifer with a cap and when the level of water flow of the river or aquifer is not sufficient to satisfy the wur that have been granted, then these wur act as shares (i.e. certain % of river flow or maximum cap in the case of an aquifer). The wur are not sector specific and can be transferred among sectors such as from agriculture to sanitation, industry, mining etc. The wur are defined as diversions which have not successfully dealt with in the Water Act. This means that non-consumptive (e.g. hydropower) use rights allow the owner to divert water from a river
with the obligation to return the same water unaltered to its original channel. Consumptive use rights do not require that the water be returned once it has been used (Donoso, 2011). Hence, the sustainability of rivers and aquifers is compromised and secondly, increased conflicts can arise with downstream users due the existence of wur over return flows. Further robust reforms need to be pursued in the future especially in the form of collective actions where the users between the different river sections need to agree with rules that can be beneficial for everyone and ensure the well-function of the whole river system (upstream and downstream).

Delving into the definition of water rights, according to the resource availability, they are divided into permanent and temporary and according to the time of the use of the resource the rights are classified into continuous, discontinuous and alternated. Also, depending on the use of the flow, consumptive (irrigation) and non-consumptive (hydropower) water rights are also defined (Rey et al., 2012). The WC 1981 did not address any environmental sustainable policies (third-party effects and environmental impacts) except in 2005 when it was reformed to consider regulation for the establishment of minimum ecological flows. However, the registration of water rights was not adequate. There is a significant number of rights with no record although they are in use and exercise. The lack of legal certainty of water rights and the absence of a system to identify the current right holders in a given watershed or river section are the main difficulties that authorities have to face to make efficient functioning of water markets in the country (Rey et al. 2012).

Moreover, the 1981 WC clearly separated the role of the government from the private sector. As far as the Government Bodies are concerned, the role of State in water management is mainly focused on measuring and determining the availability of water resources and on protecting natural resources, the impact assessment service and environmental legislation. The Directorate General of Water (Dirección General de Aguas, DGA) is responsible for granting, monitoring and enforcing of wur. It has very little regulatory authority over private water use and can’t cancel or restrict water rights once
they have been granted (Rey et al. 2012). Finally, the management of water in day to day decisions and issues is carried out by the User’ organizations. They are three types of User’s associations, the first one operates on natural resources, rivers, and aquifers, the second one is responsible for the distribution of water in channels and the last one is water communities which are collectively managed by users that do not have sophisticated channels. These organizations do not own water rights, however, they have arbitration powers and represent members against third parties. Thus, any entity holding water rights must join any organization or association established in the Water Code (Rey et al., 2012).

Empirical evidence in assessing the efficiency in water markets in Chile indicated that in terms of volume traded remained limited but reallocation has performed reasonably well, even though third party effects and speculative behaviour reduced efficiency (Mole and Berkoff, 2009). Grafton et al. (2010) pointed out that 8 to 32% of the agricultural sector’s contribution to regional GDP, $22 million annually, is attributed to water markets. However, there is an uneven spread of pricing information in the market that particularly disadvantages market participants with fewer resources and also increases transaction costs. Donoso (2012) suggested that the allocation framework based on a market allocation system established by the Water Code in 1981 has been efficient from an investment point of view, as several economic sectors undertake significant investments to improve water use efficiency and to increase the availability of groundwater through exploration. Likewise, the free transaction of water use rights, even though in many areas water use rights markets have not been very active, constitutes an efficient reallocation mechanism which has facilitated the reallocation of granted rights (Donoso, 2012).

Although the Water Act in 1981 defined the rules of the game (Level 2 institutions) and its subsequent amendment in 2005 to consider regulation for the establishment of minimum ecological flows (Level 3 institutions), their implementation was not effective.
Therefore, we can’t consider the Chilean water market as robust due to the lack of proper registration of water rights, high transaction costs or conflicts between upstream and downstream users over the return flows, with the exception of unbundling the rights from land. Therefore, the effectiveness of water markets in Chile could be enhanced by employing more robust changes which could overcome the following difficulties. Firstly, data on wur transactions and prices for buyers and sellers is needed to overcome the lack of wur and wur market information. Secondly, wur needs to be clearly specified, ownership secure and formally registered. Thirdly, the existence of transaction costs can be dealt with collecting information on water transactions, water right prices and water market activity. Finally, a rapid, efficient controversy resolution system to solve conflicts among water users needs to be further developed (Donoso, 2011).

3.4 Water markets in Spain

This section evaluates the performance of water reforms to facilitate inter-sectoral water transfers in the Tagus river basin in Spain based on the study by IMDEA (2011) where Williamson’s four-levels of institutions is implemented. An overview of the legal, institutional and environmental barriers to water markets in Spain is also provided in this section.

Water use rights are defined by the abstraction point, type of use, calendar, plots and crops to be irrigated and irrigation technologies, usable volume or flow and return flows (Garrido et al. 2012). The type of use, location, abstraction or return points cannot be changed without an explicit approval by the River Basin Agency (RBA) (Rey et al. 2012). Rights differ in the priority of their access to water depending on the type of use (domestic, environmental, agricultural, hydropower or industrial) (Rey et al., 2012, Calatrava et al. 2012). During the drought events in Tagus river basin, in 1993 and 2002, two water transfers occurred, from irrigators to urban suppliers. In the first case, water was transferred from irrigators in the Henares Canal to provide drinking water to several towns supplied by the Mancomunidad de Aguas del Sorbe (MAS) (Sorbe Water
Community) (with Alcalá des Henares being the most important town). In the second case, water was transferred to the city of Madrid from irrigators of the Alberche river.

Both water transfers addressed the fourth level of Williamson’s institutions as their aim was to allocate water from low to high value users. These water transfers were allowed thanks to the reform of the Water Law in 1999 (Law 46/1999) which introduced the so-called water right lease contracts (contratos de cesión) and water banks (centros de intercambio) that eased certain transfers of water rights for a given period of time including a pecuniary compensation (IMDEA, 2012). However, in the case of transferring water from the irrigator areas of the Alberche river to the city of Madrid, no clear and well-defined registration of the irrigators water rights was available a priori. As a result, not all of the farmers were able to participate in the trading process. In 2005, the government with a decree (RDL 15/2005) allowed water users adjoin to public irrigation land to sign transfer contracts provided some conditions were met (IMDEA, 2011). The 1999 and 2005 water reforms can be considered as changes on level 2 and 3 institutions (Williamson), as the “formal” rules (Institutional Environment) and “play of the game” (Governance). Also, the 1999 water reform defined that water transfers need to be approved by administration which can be time consuming, up to two months. Moreover, the river basin authority can reject a water transfer if he concludes that negative impacts on the environment and water resource might occur. With respect to the drought events in 1993 and 2002, the Ministry of the Environment, Rural and Marine Affairs (MARM) and the Ministry of Agriculture, Fishing and Food Affairs involved in the regulatory process (IMDEA, 2012). This legal reform addressed level 3 institutions with the government playing an important role in the whole process, though concerns about high transaction costs might rise.

Although these legal reforms attempted to define the rules and play of the game to facilitate the implementation of water trading in Spain, they can’t be considered as robust since there are still considerable barriers to trade, which Garrido et al. (2012) split into
legal, institutional and environmental. The legal barriers include market barriers, e.g. the number of buyers and sellers and barriers related to the definition of water rights e.g. rights to consumptive uses cannot be sold to holders for non-consumptive uses (hydropower) and vice versa. Institutional barriers include regional and intersectoral barriers occur when representatives of one sector collectively fights exchanges that go against its political standing within the hierarchy of water rights and political priorities (Garrido et al. 2012). Finally, as far as environmental barriers are concerned, these are those enforced by public agencies responsible for the ecological quality of rivers and water bodies. For instance, the minimum environmental river flows, are based on modeling evidence, which are hardly contested (Garrido et al. 2012).

Rey et al. (2012) and Garrido et al. (2012) give a comprehensive overview of the water market activities in Spain, which include informal trading of surface water resources, trading of private groundwater rights, formal lease contracts, purchase of land to use water in other parts of the basin, inter-basin water trading and public exchange centers/water banks and option contracts. Also, IMDEA (2012) provides empirical evidence about the water trading activities from one basin to another such as in the Tagus-Segura interconnected basins, emphasizing that there are significant legal restrictions and they are mostly limited to emergency periods. The following section provides a list of robust recommendations for improving the water trading schemes in Spain and a new robust governance model for sustainable water resource and use for this country.

4. Policy Recommendations

We believe that this study can be a valuable roadmap for understanding which factors weakened or strengthened the development of water markets, and how they could be further developed to be environmental, social and economic accepted. Therefore, combining together the lessons learned from the Spanish and international experience from the evolution of water markets where different legal systems exist, we provide
robust recommendations for improving the water trading schemes in Spain. These include: 1). Climatic, geologic and hydraulic information for the definition of water rights; 2). Registration institutions to record water rights; 3) Unbundling water allocation and management regime; 4) Flexibility in water transfers (use or rights), 5) Recognize the environment as legitimate water user e.g. establish guidelines for minimum environmental flows (combination of robust modelling approach (see Katz, 2012 for more detail) and stakeholder consultation), avoid external effects on third parties e.g. return flows or over-allocation problems.

The definition of secure water rights allows for the development of water markets. Clear information of how much water is allowed for abstraction, use and minimum ecological flows, within a defined period and location and registration of these rights increases transparency. An unbundling allocation licensing regime allows changes in water reforms to be target specific. In other words, each component is defined in a manner that enables decisions about one component to be made without consideration of implications for other components because each component is defined in a hydrologically and legally robust manner (Young, 2012). Unbundling, coupled with verification of registers, could therefore maintain transactional costs low and enable water trading.

Moreover, freely water transfers, from one sector to another, for instance from agriculture to urban, industry and energy, and between non-consumptive and consumptive uses could facilitate water trading from low to higher-value uses. For instance, the water reform in Andalusian Water Law in 2010 allowed changes in the priority system, meaning that irrigators are on the same level with other users such as industries and therefore, exchanges between these users are permitted (Garrido et al. 2012). Another example is illustrated by Gomez (2012). In the case of Mallorca, the possibility of the water supply firm to buy rights from farmers in dry years is showed to avoid the cost of infrastructures such as dams and desalination plants required to secure
the supply of drinking water. This way water trading allows supply security with lower water tariffs and an income guarantee for farmers in dry periods (Gomez, 2012).

Furthermore, environment is an important user in the whole process. As Gomez et al. (2012) emphasized transferring water from one source to another may have external effects on third parties coming from the fact that farmers in the low part of the basin use the return flows of farmers and other users utilizing the water upstream, including recycled wastewater and/or discharged cooling water from power plants, which might be essential to maintain water flows in the river. Therefore, recognizing the environment as legitimate water user could result in environmental sustainability of the water resource and in avoiding externalities.

Other recommendations equally important include: 6) Reduction in institutional and monetary transactional costs, for instance the process of approval of water trading and the establishment of uniform fee across basins for registration and approval fee of water trading, 7) Registration institutions to collect information and data from trading activities, for instance the amounts of transfers can be collected and be classified by size and nature of seller and buyer (e.g. agricultural to urban or agricultural to agricultural). As a result, the perception of the process, the quality and accessibility of market information and guarantee of market proficiency are enhanced, 8) Cost-benefit analysis, i.e. to quantify benefits and externalities, 9) Establishment of specialist environment courts responsible for resolving disputes concerning water rights which do not have to be located solely at the level of the water administration but specific local resource management bodies such as water user associations (Hodgson, 2006). As the users participate in the administration of water resources, their expertise in local issues may effectively influence the development of the water market and fairly resolve any disputes which may arise (World Bank, 1999). 10) Water markets can be particularly successfully if they are localised, meaning that the new institutions can easily adapted to the specific circumstances of the region and moreover, if they are conducted among homogeneous groups as they share the same
concerns and needs, and their expertise can be important for dealing with local disputes and environmental sustainability of the water resource.

In addition to the above robust recommendations for improving the water markets in Spain, a new robust governance model is proposed. This model is based on Sharma’s (2012) approach but re-prioritizing its components. A robust governance model for Spain is depicted in Figure 4 and should include the elements of environmental responsibility, political support and action, social education and acceptance, effective administrative systems and adaptive governance. Environmental responsibility implies that environmental guidelines, in relation to how much water is diverted, abstracted or return to a water resource, needs to be clearly defined by those who have the knowledge to do so e.g. environmental scientists. Political support is required in the case when water reallocations are not beneficial between parties (sectors) or when environmental guidelines are not respected. Together with political support goes political action which can take the form of social education programs, design and enforcement of legal rules and establishment of administrative institutions (Sharma, 2012). Social education and acceptance implies raising awareness of the environmental, economic and social value of a water resource for both stakeholders and citizens. Effective administrative systems can take the form of not only recording information related to market activities but also of a dispute resolution mechanism in case voluntary agreements are not successfully (Sharma, 2012). Finally, adaptive governance implies that institutional regimes need to be flexible to meet unpredictable conditions, i.e. to be able to change the rules of the game, for instance to include new scientific knowledge or to apply a variety of policies in the face of changing conditions (Sharma, 2012, Walker et al. 2002, Drieschova et al. 2008).
Therefore, as it can be seen from the figure above, the role of the political component, to re-define the rules of the game, is given the highest priority and second priority is given to environmental, economic and social needs. Therefore, legal and political institutions that support clear property rights, recognize the environment as legitimate user of water, encourage support and action, will lower the transaction costs of trade and facilitate the smooth exchange of water from low to higher-value uses, facilitating therefore the realization of environmental, economic and social objectives.

5. Conclusions

In this paper we reviewed the introduction and development of water markets, based on the new institutional economics approach, in countries such as in Australia, USA (California and Colorado), Chile, and in Spain. Based on the theories by Williamson, North, Coase and Ostrom on transaction cost economics, property and collective actions, we reviewed the evolution of water reforms changes and identified factors that strengthened or weakened the development of water markets. Based on the lessons learned from the Spanish and international experience review where different legal systems exist, we provided a list of robust recommendations for the improvement of water
markets in Spain, a country in which the absence of robust water governance and effective surveillance does not allow water markets to be efficient and socially accepted (Garrido et al. 2012).

The main results from the international review on the evolution of water reforms for water markets suggest that well-defined rights, appropriate regulation of water markets and changes in beliefs and notions are preconditions for the development of water markets. Additionally, institutional representation of the environment is of paramount importance and needs to be included in the robust design principles for sustainable water resources and well-function of markets. We believe that the international review on water markets can be a valuable roadmap for understanding why water markets function or not and how they could be further developed to fulfil environmental, social and economic objectives. Therefore, a list of robust recommendations for improving water markets in Spain includes among others not only the definition of secure water rights, for instance, registration of rights or recognition of environment as a legitimate user, but also monitor of the water trading activities, for instance collection of information for prices and quantities or cost-benefit analysis for quantifying benefits and externalities. Moreover, it is concluded that water transfers can be particularly successful if they are localised, meaning that the new institutions can easily adapted to the specific circumstances of the region and if they are conducted among homogeneous groups as they share the same concerns and needs compared to heterogeneous group. Their expertise can also be important for dealing with local disputes and environmental sustainability of the water resource. Finally, following the approach of Sharma (2012) the well-function of water markets in Spain would further require the establishment of a robust water governance model in which environmental responsibility, political support and action, social education and acceptance, effective administrative systems and adaptive governance are important components. The highest priority is given
to the role of legal and political institutions and second priority to environmental, economic and social needs.

We hope that the framework presented in this paper will function as a tool for researchers and policy makers in Spain and other European countries to understand how water markets can be further developed to be economically and environmentally efficient, and socially accepted.
References


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1. Introduction

With 71,000 km² (~24% of the state territory), the Po river basin is the largest (single river) basin in Italy and the economically most important area. The basin area is home to 17 million inhabitants (~28% of the state population). More than one third of country’s industries producing 40% of the national GDP are located in the basin area. The agricultural output accounts for 35% of the national production. The agricultural sector generates an added value of about 7.7 billion €/year (~1.2% of the total added value produced in the basin). The one thousand or so hydroelectric plants installed on the Po River and its tributaries generate on average 20 billion kWh/year (~48% of the installed hydropower in Italy). Additional 400 thermoelectric plants generate around 76 TWh every year. The natural and artificial lakes in the basin regulate a volume of 1,858 million m³ per year (AdBPo, 2006).

The river basin spreads over eight (out of twenty) Italian regions including Valle d’Aosta, Piedmont, Lombardy (all three entirely included in the basin area), Emilia Romagna (with about a half of the area included in the basin), Autonomous province of Trento, Veneto, Liguria and Toscana (marginally included in the basin area).

![Po river Basin: Administrative borders](image)

Figure 1 - Po river Basin: Administrative borders

Po river basin annual average precipitation is 1,108 mm with maximum values in the Alps, over 2,000 mm per year and minimum in the eastern Paduan plain, 700 mm per year (AdBPo, 2006). This amount of precipitation produces an annual water flow of 78 billions m³, which correspond to a water flow of 2,464 m³/s. Two third of this flow runs on
the surface, that is approximately 47 billions m$^3$ per year, 1,470 m$^3$/s. The remaining 31 billions m$^3$ are consumed by evapotranspiration and deep percolation. Two mountain chains, Alps and Apennines, feed all rivers in the basin. River cycle characteristic depends on the source of water. Alpine rivers have water flow peak in summer due to ice melting, while Apennines’ rivers have lowest peak in summer due to their dependency from precipitations, and highest peaks in spring and autumn.

Po river basin is water rich thus its surface water component is remarkable. The principal reticulum includes 141 major water affluents (>20km of length), while the secondary surface river network is nine times more extended than the primary river network, which lengthens in the basin for over 6,750 km (AdBPo, 2006). Artificial networks, including irrigation channels and drainages, are also highly developed throughout the basin. This complex and extended water network is the result of thousands of years of human alterations of the natural environment. Flow of water from mountain basins and natural lakes to the Po river running along the Paduan Plain is intensively interfered by artificial abstractions, rice field submersions, dripping irrigation, deviations for irrigation channels, irrigation losses, and the interaction between surface water with aquifers. The surface water network also includes major artificial irrigation canals. Among them the Cavour Canal, the Emiliano-Romagnolo Canal (CER) and the Muggia Canal are of the most important in terms of water flow derived from the natural network.

Figure 2: Po river basin major surface water bodies
2 Water demand management for drought events: existing policies

2.1 Institutional instruments

This section discusses governance policies that are already in place to deal with drought events in the Po river basin. Governance policies are divided between ordinary and emergency. Ordinary policies include the adoption of several plans such as management, water protection, area water and hydrological plans. Agreements among several authorities to ensure sufficient water supply for water uses in case of droughts, like the protocol of Intent, are also discussed. Actors that are responsible for the water related risk governance are also included in this section. Finally, the management of emergency includes emergency plans and actions to respond and deal with a disaster.

2.1.1 Ordinary Governance

The actors that are responsible for water related risk governance include the state government, River Basin Authority, Irrigation and Land Reclamation Boards, water supply and sanitation authority, civil protection, steering boards of regulated lakes and water utilities.

The state government is split into three administrations, regional (RA), provincial (PA) and municipal (MA) administrations. Each administration develops and adopts different plans for the management of disaster risk and preservation of water resources. For instance, RA develops the Regional Territorial Plan (RTP), which is a complementary plan for the Provincial Territorial Plan and General Urban Plan, developed by PA and MA respectively. Moreover, the River Basin Authorities, being public bodies, are responsible for hydro-geological risk, integrated resource management and environmental protection conservation for the river basin district areas. Irrigation and Land Reclamation Boards are also public bodies, which represent the interests of farmers and their concern is on costs
and benefits of irrigation, transportation and distribution of water for irrigation purposes. The Optimal Territorial Area Authorities (AATOs) are regulatory bodies with tasks related to long run economic planning and control activities. These authorities regulate the water utility that operates in each Optimal Territorial Area (ATO) and is responsible for the provision of both water and sewerage services. Finally, the Steering boards of the regulated lakes are responsible for the hydrologic monitor of rivers, lakes and precipitation in their respective basins.

Since 2006, the territory of Italy was divided in eight Hydrographic districts responsible for the water management of the river basins. Each river basin adopts its own district River Basin Management Plan (RBMP). The RBMP specifies the necessary infrastructure and investment projects to deal with extreme events such as drought and flood and to ensure the protection and conservation of both water quantity and quality. Within each river basin, as part of the RBMP, additional plans are adopted such as management and water protection plans, area water and hydrological plans. The management plan is focused on monitoring the status and the compliance with the environmental objectives for surface and groundwater only. The water protection plan, being more general compared to the management plan, includes actions, which are of great importance in order to protect water resources. The water protection plan, prepared at regional level with the collaboration of Basin Authorities, Regions and local authorities, deals with all the aspects of water quantity and quality, surface and groundwater, control of emissions and compliance with environmental objectives. Furthermore, the area water plan is another instrument for the management of water. It is prepared by the Optimal Territorial Area Authority and provides the guidelines for planned and future investment projects that can be implemented in an area for the efficient and sustainable use of water. The hydrological plan, prepared by the Basin Authorities, defines structural and non-structural actions for the protection of soil and water uses. It mainly concerns with the
identification of areas to critical exposure and risk (actual and potential) to natural
disasters, the action plans to deal with the damages due to natural disasters and to protect
soil and water. Finally, except for the adoption of several plans within each river basin to
deal with water crisis events, agreements between different authorities to guarantee
sufficient water supply for water uses are also feasible. One example is the protocol of
intent that was implemented during the 2003 drought event, with the coordinated and
integrated intervention of several authorities. The agreement was related to the release of
additional water from hydropower reservoirs and reduction of agriculture withdrawals to
compensate the reduced water discharge of the Po river, and maintain its level above the
minimum environmental flow required by law.

2.1.2 Emergency Governance

The National System of Civil Protection is responsible for the management of
emergency when a major natural disaster event occurs. Civil Protection responsibilities
include all activities required to safeguard the integrity of human lives, goods and the
environment against the damage and hazards deriving from natural disasters and
catastrophes. Its administrative sectors have different powers and functions in the
forecasting, prevention, planning and management of emergencies. The declaration of
emergency results in the temporary suspension of any ordinary governance and its
management is assigned to a commissioner, which is chosen among institutional and
political representatives with acts and competences only for the duration of emergency.
Furthermore, each region, province and municipality adopts emergency plans to respond
and act in case of disaster events. Except for emergency plans, other instruments that can
assist in monitoring and alerting for disasters include the establishment of emergency
operational or hydraulic prompt intervention centres. Regional Civil Protection Agencies
are also part of the system. One example is the Region of Emilia Romagna (RER) Civil
Protection Agency whose responsibility includes the declaration of emergency at regional
level and the release of emergency funds for affected areas.
2.1.3 Existing adaptation policies

Adaptation policies are still not well rendered in Italy. In order to identify specific climate adaptation related policies, one should deduce them from existing instruments and on-going processes of the water management system reform. For example, the recent effort of reorganizing the National Civil Protection, which aims to reduce the total number of “state of emergency” and use the ordinary governance instead, could be seen and an effort to adapt governance to more frequent disasters and extreme events. The Water Management Plan (Piano di Bilancio Idrico), which includes a Drought Early Warning System, and the coordinated management of the whole water district in terms of flows and withdrawals from all sectors, is an effective measure under development by the River Basin Authority that amongst other issues will tackle climate change processes. Within the European Water Framework Directive (WFD) the creation of new District Water Authorities is still an on-going process. Italian Territorial Authorities (ATOs) reforms aims to the same objective. Both examples could indirectly be related to climate change adaptation actions, even though it is not possible to identify them as such specifically.

Instead the recent National Adaption Strategy (NAS), currently under elaboration by several institutions and coordinated by the Euro-Mediterranean Centre for Climate Change, is the first national effort to define climate change adaptation strategies, which will define specific adaptation policies.

2.2 Economic instruments

This section will discuss the economic instruments that are adopted in the Po river basin to deal with droughts. Economic instruments include water tariff in urban water supply and agriculture, whereas financial instruments include national irrigation plan, cohesion policy, rural development and social protection. The water tariff system in Italy
was introduced in 1996 and since then it has been subject to several modifications. One of them occurred in 2006 when a return in the capital employed was introduced for the calculation of water tariff formula and was fixed at the level of 7 per cent. The purpose of this policy was to ensure that the water utilities have sufficient revenue to make investments in maintaining and enhancing the capital infrastructure, in improving the quality of service, e.g. unplanned disruption and environmental performance and reduction in the water losses. Another modification in the tariff system occurred in 2008 in the Region of Emilia Romagna with the introduction of performance indicators. Water utilities are allowed to increase the prices charged to consumers if water efficiency goals are achieved, e.g. reduction in leakages, improvement in the quality of service. On the other hand, if water saving targets are not accomplished, then water utilities are penalised by reducing their tariffs. The empirical evidence so far showed that over time on average there was a declining trend in water abstraction levels and in distribution losses, even if the latter still remains at high levels. Investments to improve the capital infrastructure and environmental performance, however, led to high increases in water and sewerage bills. Therefore, policies aimed at improving the affordability of water services for groups that are most in need resulted in the introduction of innovative water tariff structures (W2Adapt, 2012). For instance, the ATOs of Bologna, Ferrara and Parma introduced the social tariff water system to help low income households. The subsidised tariff system, based on the number of household’s members and the level of consumption, ensures social equity for larger families, and single consumers are encouraged to reduce their consumption (EPI-WATER, 2011). ATOs are currently under reform, towards a Regional body, which will regional water integrated systems and will probably shape future policies.

As far as the water tariff system in agriculture is concerned, the system of payment is rarely measured on water flow but on cultivated surface, which can result in over-allocation water and inequity problems between irrigators and non-irrigators. A policy aimed to overcome the above difficulties was the introduction of volumetric pricing
system for irrigation use in a small area, called Tarabina, which is located in the Province of Ravenna (W2Adapt, 2012). The new voluntary volumetric pricing scheme with the agreement of both irrigators and non-irrigators included three new components. Firstly, a fixed component representing maintenance and operation (M&C) costs payable only by non-irrigators and two volumetric components payable only by irrigators based on water use and other M&O costs. The empirical evidence so far suggests that firstly, there was a substantial decline in tariffs for non-irrigators and secondly and most importantly, volumetric price system for irrigators improved water allocation among farmers and overall water uses, providing incentives to use less water for farmers with lower marginal value of water (EPI-WATER, 2011).

2.3 Financial instruments

Several financial related policies are also in place to manage and improve the efficient use of water in agriculture such as National Irrigation Plan (NIP) and National Solidarity Fund (NSF). The first is related to infrastructure and irrigation projects such as structural and technological improvement of the irrigation networks and actions targeting to enhance resource management through interconnections. In the National Irrigation Plan each region specified the level of investment for the maintenance and restoration of irrigation systems and different set of interventions. Overall, the highest level of investment is reported for the areas of Emilia Romagna, Piedmont, Umbria and Lombardy, whereas the lowest level of investment occurred in Valle d’ Aosta, Bolzano Autonomous province and Liguria. The rate of completion of the planned works reached the level of 100% in the region of Bolzano Autonomous province, Marche and Valle d’ Aosta, whereas the lowest levels occurred in Tuscany and Umbria (W2Adapt, 2012). The National Solidarity Fund is related to the promotion of actions to deal with damages to agricultural and animal productions, to farm infrastructure and equipment, in the areas
affected by natural disasters and extreme events. The purpose of its reform in 2004 was the inclusion of more insurance services such as the increase in the maximum level of national contributions (80%) for insurance contracts with damage threshold at 30%, multi-risk insurance contracts for crop yields and national insurance plan, denoting the amount of national funds and the parameters to calculate the contributions (PREEMPT, 2012). In 2006 four Regions in the Po river basin, namely Veneto, Piedmont, Lombardy and Emilia Romagna filed for compensation from the fund, which covered 5 per cent of declared damages in each region. Also, in 2007 Emilia Romagna and Veneto appealed to the fund again, which covered 3 and 10 per cent of the damages, respectively. The empirical evidence so far suggests that the releasing found covered 1.74 per cent of the recognized damage over the period 2000-2010 (W2Adapt, 2012).

The Common Agriculture Policy (CAP) aims to ensure the economic sustainability of agricultural sector in the EU is the main EU financial source in the Po river basin for two main policies ‘pillars’, market aids and rural development policy. Market aids were regulated by Common Market Organizations (CMO), which was formed to ensure stable agricultural and to protect domestic production and trade through the imposition of import tariffs. Market aids included several types of plant products such as cereals, fruit and vegetables, sugar, rice, cotton, bananas, potatoes and several types of animal products such as milk and dairy products, beef and veal, sheep, goat and pig meat. With respect to the rural development policy, it is accomplished by the financing of measures that favour a suitable rural development through local (regional) Rural Development Programmes (RDPs). These measures are of great importance for the protection and conservation of environment and the security of farmers’ incomes. These measures can vary from agricultural resources management, improvement in the agriculture infrastructure to compensation from production losses due to damages by natural disasters.

The CAP has undergone several reforms in the past in 1999, 2003 and 2007, which resulted in changing the magnitude of contributions from CMO and rural development policies for agricultural aids in the European countries. These reform included cross-
compliance direct payments to ensure environmental and agricultural protection e.g. protection of soil and water and avoid the deterioration of habitats. The next CAP reform focused on providing more financial support to the promotion of rural development policies (Pillar II) than markets aids (Pillar I), whereas the latest reform introduced a green payment to encourage the adoption of agronomic practices by farmers. The empirical evidence for the Po river basin over the years 2003-2007 underlines that market aids decreased from 73% to 59%, whereas the EU financial support for rural development policies substantially increased from 24% to 40% (W2Adapt, 2012).

3. Future trends in water availability and uses in the Po river basin

In the recent past in Po River basin, the worst drought events occurred in the last decade (2003, 2006-2007 and lately 2012). Climate change and land use changes are strongly influencing water security in the basin. The IPCC 4th Assessment Report (AR4) includes the Po valley among the European continental zones affected by a shift of rainfall regime and amplified extreme events (Naldi et al., 2008). The future changes in precipitation in the area are less clear. All models show a northward shift of the rain bands but the Po river basin is located in the transition zone that is rife with substantial uncertainties. It is expected though that increased temperature will strongly influence the liquid/solid precipitation ratio and the underlying retention effects in the water runoff. According to Coppola and Giorgi (2010) climate change signal over Italy varies seasonally, with maximum warming in summer and minimum in winter season. According to the models precipitation will decrease substantially in summer and will increase slightly in winter. Seasonal temperature anomaly probability functions (PDF’s) highlight a shift to higher temperature coupled with a flatter and broadening both in temperature and precipitation patterns, especially in summer. This implies increased extreme climate
events, such as very dry seasons and extremely hot seasons (Coppola and Giorgi, 2010). Other studies from Im et al. (2010) about the local effects of climate change over the Alps, shows a consequent effect of temperature increase, with decreasing snow cover, evapotranspiration increase and decreasing soil moisture in spring and summer. The models developed by Im et al. (2010), suggest an earlier peak snowmelt season, which coupled with lower snow accumulation, reduces the water retention capacity of the basin. This fact increases water insecurity to warmer summers. Concerning the trends of river water discharge observations, the Euro-Mediterranean Centre for Climate Change (CMCC) analysed river discharge extremes of the Po river. Conclusions suggested that the test performed on minimum and maximum discharge with a confidence level of 5 per cent, show that the series are homogeneous only within the period 2023-2002 (Vezzoli, 2012) for minimum discharge. In the last decade the study identified a sharpening in minimal discharge, which is the only signal of climate change in the dataset, but it also highlighted that the issue needs further investigation, because the period 2003-2008 recorded two of the most severe droughts (in 2003 and 2006-7) ever occurred in the Po river, which could undermine the statistical trend. According to this study river flow discharge are quite uncertain. Climate change effects have also been investigated by ARPA Emilia Romagna (ARPA-ER), which focused on the water budget of its region. Even if the results are characterized by high degree of uncertainty, it is worth to report that ARPA-ER recorded that Emilia Romagna mean temperature increased by about 2°C (~ 0.5°C/10y) over the past forty years (Cacciamani et al. 2010a) and precipitation decreased by some 20 per cent (Cacciamani et al. 2008). ARPA-ER reported that the intensity of single rainfall events increased whereas the number of the rainfall events decreased. With almost 50 per cent decrease from the previous long-term average, the decline in average precipitation is particularly pronounced in spring and summer periods. Snow cover and volume of glaciers show similar trends, as a consequence of shorter snow accumulation seasons. Cacciamani et al. (2010b) stated that the precipitation drop started in the early ‘80s. The difference in mean rainfall quantities in the last 25 years is estimated to be about 100 mm.
It is a noticeable measure, corresponding to 10 per cent of the mean annual rain in Emilia Romagna. However, these conclusions disagree with the analysis complied by CNR (Nanni et al. 2007). This research focus on secular time series measured in 100 Italian stations. The decrease in precipitation is still visible, but it is defined as “minor and statistically little significant”. Concluding, the uncertainty about climate scenarios for the Po River Basin is still great. Although several studies have reported changing trends in precipitation and river discharge, the issue requires further investigation to confirm a specific trend. Due to the geographical position of Po River basin in the climate change transition zone North-South and due to its orographic characteristics, influenced by the Alps, the future climate scenario is still extremely uncertain. However, based on CMCC (Gualdi, 2012) estimations, a climate change seasonal baseline scenario, could be defined as follows: 1) Lower precipitation in all seasons, except for Winter; 2) Higher temperature in all seasons; 3) Higher evapotranspiration in all seasons.

Concerning water uses, this study defined a “most likely “socio-economic scenario, which is used to estimate the water balance of the basin. The scenario considers all sectors, from civil supply to agriculture and industry. The first analysis concerns the “Demographic Development scenario” (ISTAT, 2011). The Northern part of Italy, which for the most part is covered by the Po river basin, will see a moderate demographic grow, around 10 per cent but a quite consistent transformation of its social distribution, which foreseen a shift of the socio-demographic fabric towards aged and non-Italian resident population. It is estimated that these factors could induce new water demand patterns of domestic consumption. In relation to the industry water demand, it could be considered that the industry production delocalization process is now stable. Therefore there is no addition water demand foreseen for the industry sector in the basin. According to the new National Energy Strategy, currently under development, the sector will not change drastically in the next future. The renewable production trend will further increase, with a
specific focus on biomasses, which could indirectly, through the agriculture sector, induce additional water demand. However, based on report, the industry sector water demand could be considered stable in the medium-long term. Agriculture is instead another matter. Sector water demand trends are of extremely difficult interpretation. European Policies, such as the Resource Efficiency Flagship of the EU 2020 Strategy calls upon Member States to increase irrigation efficiency. Rural development funds, the new European CAP 2014-2020 and the National Irrigation plan include consistent efforts for enhancing irrigation efficiency. However, as highlighted by ISTAT in the recently issued Sixth National Agriculture Census (ISTAT, 2011), the sector faces long lasting contraction.

Therefore based on this assumption the “most likely” scenario developed in this study for the Po river basin foreseen: 1) Industry and energy production sector’s water demand will not see any consistent change in the medium-long term. The new National Energy Strategy does not change significantly the Italian energy mix. Even though renewable energy will increase, water demand change for energy production will be marginal; 2) Domestic water demand will most probably show a slightly increase due to socio-demographic changes in the short-medium term. In the long term the tendency is stable or even decreasing because of the implementation of water losses reduction measures, building efficiency and conservation promotion; 3) Water demand for agriculture production shows opposing trends. Irrigation efficiency will optimize the system, inducing possibly additional production. Increasing trends in temperature and evapotranspiration will further raise water demand for food production. Increasing biomass production for energy purposes will augment the stress on water resources during drought seasons. Diffusion of water resilient crops will further decrease the water demand of sector; 3) Water availability within the Po river basin will be no longer abundant and easily accessible. Reduced snowing precipitations and glacier melting will affect the river discharge patterns influencing the seasonal water cycle. Uncertainty in future river water discharge is still great. Saline intrusion will further worsen the fragile condition of Po river delta; 4) Climate change will further increase meteo-climatic inter-
annual variability. Extreme climate events, such as very dry seasons and extremely hot periods will affect more frequently the area.

4. Proposed Water Demand Management (WDM) policies

The analysis of the Po river basin case provides the opportunity to draw a proposal for the implementation of Water Demand Management policies, such as water efficiency in agriculture, civil supply and energy sectors, innovative economic instruments, like water tariff systems, water abstraction rights, water transfer and trading, and finally water governance improvements.

4.1 Reforms implementation and coordination at RBD scale

Besides being fragmented, the Italian legislation concerning water management is constantly under radical reform. The continuous reform of the water management system, even if inspired by optimization purposes, created a muddy institutional that weakens efficiency.

There are five main issues that undermine the efficiency of governance at RBD level. The first concerns the installation of RBDs’ Authorities from the Water Framework Directive 2000/60/EC. The Italian territory is now divided into 8 districts managed by a District Authority, which should replace the former Basin Authority. Up to now (since 2006), RDBs’ Authorities has not been established and therefore former River Basin Authority are still in force. The second issue concerns the Land Reclamation Boards, responsible for the practical implementation of the measures regarding land and environment management and irrigation. They are currently subject to a radical process of reorganisation. The third issue is related to the Optimal Territorial Areas (ATO), responsible for the civil water management. They have been suspended and they are
going to be reorganised on a provincial and regional bases. Their responsibilities have not been transferred yet. The fourth issue concerns the “Vigilance Committee for water uses” which has been abolished and restored with different names several times in the past. At present the vigilance committee is repealed and the transfer to the vigilance committee for energy and gas is undergoing. The last issue is related to the National Referendum held on the 13th of June 2011, which abolished the 7 per cent utility “profit” on water tariff and the transfer to private sector of public water services, allowing for public-private partnerships. At present the Government is still in the process of enforcing the results of the public consultation, causing large uncertainty in the future of the water supply sector.

Po river basin district is the only district that covers the same territory, which was defined with the water reform in 1989. Despite several initiatives promoted by the River basin Authority, it is still unclear the identification of an inter-sector strategy for the water cycle management. Main issues are related to institutional conflicts caused by former laws that on one side managed and protected water resources but on the other side established competences at different administration levels.

The new reform, transferring responsibilities from Basin Authority to District Authorities, focused the decisions making process at regional level, but at the same time included new actors in the management process. During the past, the Technical Body (‘Segreteria Tecnica’) of the former basin authority (still in force) proposed useful interaction among the basin actors. Since the eighties, the Authority implemented numerous collaboration projects with local authorities, provinces and other stakeholders. For the Po basin this cohesion is particularly important because among different stakeholders there are a lot of autonomous institutions such as the Committee for the consultation for plan activity, Cities Po River Association, River contract, Council of the riverside provinces. The latter organization convenes several riverside cities and in the past it has developed collaborations with the former basin authority and activated connections with different provinces involved in the area. One of the main projects of the Council of the riverside provinces is the “Po river valley” that has been included in the
programs co-funded with national and European funds (within the National Strategic Framework for 2007-2013).

Other than local associations, the basin institutional framework includes also institutional organizations such as the Interregional Agency for Po river (AIPO), Regional Agencies for the Environment (ARPA) and the Regional company for internal navigation (ARNI). In order to enhance cooperation among the above-mentioned actors, several projects have been implemented. At the moment this experience produced only limited results but it should be considered an important step for a possible way to manage and organize the governance of the river. Moreover, based on these experiences, the Po river basin district Authority has written one of the best structured River Basin Management Plan of Italy.

Concluding, at present it seems difficult to overpass local interests and adopt an integrated approach for the management of the water cycle. The new ‘district’ approach introduced by D.Lgs 152/2006 received strong opposition. In fact, the division of the territory into district and the creation of an appointed authority raised several issues. It is a fact that the former basin authority received more responsibilities and functions compared to its operative and financial capacity. Moreover the absence of a clear enforcement instruments combined with the necessity to define a large framework of competences is producing a serious deficit in the implementation process of the reform.

The water budget, defined as the accountancy of input and output in the basin, is essential to design the sustainability of water resource management actions. Within the Po river basin and in Italy in general, water abstraction rights are poorly coordinated at RBD scale, they are not clearly defined and most of the time expired or based on centuries-old rights. In addition to this, water abstraction entitlements are competence of the Regions, sometimes delegated to the provinces, whereas the RBD Authority has only a minor role. Moreover the monitoring of quantity and quality of water in the basin is provided by regional authorities, which do not share a common platform for sharing the data. With
consideration to this limit and to the importance of the actual and future budget for planning purposes, it is strongly suggested that any large abstraction should be subject to environmental impact assessment, prepared and coordinated by the RBD Authority. Furthermore it is highly recommended the creation of a common data platform, under the supervision of RBD Authority, for the collection, validation, achievement, and elaboration of hydro-climatic data. Modelling of future water budget should include the effects of climate change and socio-economic scenarios, which shall possibly be developed at River Basin scale by national statistical institutions (e.g. ISTAT).

Water efficiency enhancement in civil water supply, agriculture and energy production is an effective policy to mitigate the impact of water scarcity and drought.

4.2 Civil water supply

As far as the civil water supply is concerned, there are several water savings policies that can be adopted to improve water efficiency. These policies vary from setting water efficiency targets in buildings to reducing water losses or adopting other water saving policies such as rain water harvesting and waste water re-use. Water efficiency in buildings can be accomplished through the implementation of product and building level policies.

Product policies for an efficient use of water in buildings include three instruments. Firstly, the voluntary labelling whose objective is to inform the consumer about the water performance of a device, for instance, eco-labelling for taps and showerheads. The second product policy instrument is related to mandatory labelling for all water products similar to the energy-using products. The third product policy tackles minimum water efficiency requirements for water-using products that would be placed in the market in the future (PREEMPT, 2012). Toilets and showerheads seem to be the water-using product to be addressed in priority, given their high shares in the residential (toilets and showerheads) and non-residential (toilets) water use patterns (EC, 2009). On average in EU, the potential savings from the use of different household technologies can be up to 50 per cent, and the
savings from the improvement in the technological performance of household devices can be up to 25 per cent (Ecologic, 2007).

Building-level policies include four schemes which can allow the efficient use of water. Firstly, the water performance rating/auditing of buildings which be accomplished through the introduction of indicators or requirements that a building needs to satisfy, concerning water performance, and covering also other environmental issues. The above scheme can be voluntary or mandatory. Thus, both approaches can promote water efficient buildings to be certified. Another building-level policy scheme is the minimum water performance requirements of buildings whose objective is to set the standards under which a building is defined as efficient enough. Certification scheme for water reuse and harvesting, collection and reuse of grey water, and collection of water from roofs is another policy that could be further pursued and lead to significant water savings. Empirical evidence for the implementation of rainwater harvesting approaches in France and UK showed that water savings can reach up to 80 and 50 per cent respectively (Ecologic, 2007).

Another additional policy which can lead to significant water savings is the improvement in the efficiency of water distribution network. The adoption of water metering/smart metering for detecting water leakages and provide detailed information about water consumption for water utilities and customers (daily and interval collection data e.g. every 12 or 24 hours) could be investigated. It has been calculated that the leakage reduction program could potentially result in the reduction of water losses by 52 per cent nationally (the national average is 32%, from ISTAT 2011).

4.3 Agriculture sector

Water efficiency in agriculture could be achieved enhancing irrigation infrastructure and technologies and application efficiency, including a change in practices
for instance from surface irrigation to sprinkler or drip irrigation with estimated water savings to 15 per cent to 30 per cent, respectively. Substantial potential water savings can also be achieved by the change of crop patterns and the use of more drought resistant crops. For instance, the switch from high water demanding crops, like maize, to low water demanding crops reduced the vulnerability to droughts. Furthermore, the implementation of new technologies for the re-use of sewage effluent such as sand filtration or reverse osmosis could lead to significant water savings up to 10 per cent and 12 per cent (Ecologic, 2007).

4.4 Energy production sector

Moreover, a sector that uses large amounts of water is the energy sector and especially the thermoelectric as its plants produce almost 80 per cent of the total electricity production and therefore use the largest amount of water compared with other generation plants such as hydropower, wind and solar plants. Traditional cooling techniques of thermal power plants are totally water intensive as they require large amount of water from rivers. The implementation of advanced cooling technology such as dry cooling, evaporative cooling and hybrid cooling can reduce the dependence of power plants from natural water resources and therefore, and can lead to reductions in water use and consumption, by up to 80 per cent (Central Electricity Authority, 2012). An economic analysis regarding the different costs of cooling systems showed that dry cooling systems can become profitable and thus can be justified economically if the cost of water is expensive and/or the cost of power is cheap (Ecologic, 2007). Other water savings measures that can be applied in the thermoelectric generation are the use of recycling of cooling water and improvement in energy efficiency of new thermoelectric plants. The former led to several projects in Latvia, Poland, Ukraine and Hungary with substantial impact in the reduction of water consumption.
4.5 Water tariff

The economic policy instruments such as water pricing operate within the boundaries limit laid down by the regulatory environment. In Italy, the eligible costs of the WSS services are determined by central government, leaving little leverage to the lower authorities. Controversial is the cost item referring to remuneration of invested capital (7 per cent according to the Normalised Method), abrogated by the 2011 public referendum, leaving space for different interpretations as for what is the role of private sector in the service provision. Empirical evidence shows that water pricing is a suitable tool for encouraging water conservation and demand management. Water is a social good whose service provision can be governed by economic instruments. The recognition of right to water as a fundamental human right is not at odds with the participation of private sector in the water service provision. The access and affordability of water can be reconciled with water pricing in several ways. In Region Emilia Romagna (RER), it is managed by social tariffs whose costs are distributed among the wealthier consumers. Alternatively, it could be managed either by income support (connected or not to water consumption), or by facilitated payments. The performance factor (PCn) introduced in RER is an effective instrument to improve the quality of water distribution service and reduce water leakage.

However, the tariff system has not guaranteed necessary investments into extension and modernisation of water infrastructures. The planned investments in water infrastructure are by far too low in order to guarantee a sustainable and reliable water services. The failed attempt to reinforce participation of public sector in WSS provision introduced a regulatory uncertainty discouraging from further investments. The water utilities will have access to external sources of finance, such as loans, only if a sufficient and reliable stream of revenue is ensured.
4.6 Water transfer and trading

Water trading is also another effective instrument for water demand management. Water is not equally distributed in the geographical space. With the technological and engineering development, the possibilities to build more and more efficient hydraulic infrastructures permitted to create complex schemes to convey water from a river basin to another. This practice is called inter-basin water transfer or trans-basin diversion. Different could be the benefits of the water transfers but enormous could be the negative side effects of the practice under consideration. However, inter-basin and intra-basin water transfer from different catchment areas could be the most effective and efficient measure to deal with water scarcity in water scarce regions- An example of an inter-basin water transfer at the Italian regional level is the water pipeline from Basilicata and bordering regions to Puglia in Southern Italy. The water transferred from the mountainous regions to the plains of Puglia is fully integrated into the public water supply, thus serving all parts of society and economy such as households, tourism, industrial production, and agriculture (Xerochore, 2009).

In the case of the Po river Basin, the transfer could be an effective solution to cope with the structural water scarcity in dry periods in the Romagna region. Especially regarding the city of Rimini, drought events impacted in several occasions the water security of Rimini urban area. The agricultural sector in the Po plain, especially in the Piedmont and Lombardy area is responsible for large part of the water abstraction in the basin. The irrigation in the area has not always been characterised by the highest technology in terms of efficiency. The Emiliano Romagnolo Canal (CER) is the infrastructure that could be used to better and more efficiently distribute the water resources among and outside the Po river basin. The strategy could be developed in order to ensure water supply and security to the urban areas of the Romagna region utilising part of the resources saved through the improvement of the efficiency in the upstream agricultural area of Piedmont and Lombardy. The economic sustainability of the measure could be ensured by the monetary compensation offered by the beneficiary municipalities,
which will pay a service. The economic flow could be also utilised for the implementation of the infrastructural investments in the upstream areas.

Concluding, water trading is a policy which could be further investigated in the Po river basin for applicability against increasing water scarcity conditions. The structure of water abstraction rights in the basin is extremely complicated. Very often water rights are based on historical motivations, licences for abstraction are renewed automatically and rights are not explicitly formalized. Agriculture is the sector using 80 per cent of available freshwater. In the sector water in managed by Land Reclamation Boards which water rights were established in the past, when climate change was not exhibiting its effects yet. Even if surrounded by uncertainty, current meteo-climatic changes show an evident trend, which require innovative and original solution. Even though water markets at the moment have no possibility to be implemented at basin scale, future and periodically monetary compensations could still be investigated for application against increasing water scarcity conditions. For example the water transfer from upstream Lombardy and Piedmont, could be compensated from downstream, out-basin Municipality area of Rimini, when needed. Moreover the existence of a water trading scheme, could be effective in compensating the voluntary inter-sector water transfer, which is often required during drought periods, such as the one implemented in 2003 during the Protocol on Intent.

European Environmental Agency stated that the objective for the next five years is the systematic co-operation between urban water managers and other professionals and the local communities to redesign water management systems which can be integrated with other city services in order to deliver sustainable water services and simultaneously, to enhance life both within and beyond the urban environment. The exploitation of all sources within a city and their integration in the urban water cycle is of great importance in mitigating climate change shocks. In this context, international experience can be useful like the development of Water Sensitive Urban Design (WSUD) in Melbourne (Australia). WSUD is associated with the integration of storm water, groundwater supply and
wastewater management at the development scale and the protection of aquatic ecosystems. WSUD has recently been adapted and implemented with success in other countries like in the city of Hamburg (Germany). Therefore WSUD can be used as a tool by water managers and the local communities for ordinary planning, which could enhance urban resilience climate change adaptation processes. Consultation with stakeholders in Ferrara and Parma, have been useful in identifying weakness and gaps. For example during the workshops it was highlighted that network efficiency is of great significance to cope with water crisis situations and meet future water demands as well. Investments in maintenance of infrastructure like the reduction in losses from abstraction, treatment and distribution of water and in protecting the existing water resources from pollution are necessary. Research on finding alternative water sources in order to reduce the dependence of only one source, surface or groundwater, needs to be further pursued. For instance, in Ferrara 76 per cent of total potable water comes from rivers and 24 per cent from groundwater. The opposite case exists in Parma where the major source of water comes from groundwater. The construction of floating drafts in the rivers to ensure sufficient water supply, in the case of Ferrara or the diversion of water from other rivers, in the case of Parma, are considered as examples of prevention measures to reduce vulnerability of cities in extreme events. Except for management interventions, public awareness via education and information campaigns for the efficient use of water needs to be further pursued. It was finally concluded that rigorous actions towards a more water sensitive behaviour and management, such as the WSUD approach, should be taken not only during and in aftermath of emergencies, but also as a risk prevention and preparation measures to extreme events, even in otherwise water-abundant regions.
5. Conclusions

The purpose of this report was to provide the reader with a description about the existing instruments adopted in the Po river basin, Italy, to deal with drought events. These instruments include institutional, government policies which are divided between ordinary and emergency such as the adoption of several plans (i.e. management, water protection, area water and hydrological plans) or agreements among several authorities to ensure sufficient water supply for water uses in case of droughts, like the protocol of Intent. Economic instruments include water tariff in urban water supply and agriculture, whereas financial instruments include are related to irrigation plan, cohesion policy, rural development and social protection. Several financial related policies are also in place to manage and improve the efficient use of water in agriculture such as National Irrigation Plan (NIP) and National Solidarity Fund (NSF).

After defining the “most likely” scenario with respect to the water demand for the sectors in the Po river basin, after taking into account climate change, a list of water demand management policies are proposed to improve drought management. These recommendations include water efficiency in agriculture, civil supply and energy sectors, innovative economic instruments, like water tariff systems, water abstraction rights, water transfer and trading, and finally water governance improvements.
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Research Task 4.2 Outputs 3 and Output 4
EPI4Drought:
O.4. Water Scarcity and Drought – Outline of policy tools and recommended actions at the EU and MS level.
O.3. Understanding the Costs of drought damages

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O.3. Understanding the Costs of drought damages

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1. Background

The Tagus Segura WP4 case study of the EPI-water project explores the application of different economic instruments in isolation or combined as part of a policy mix to help manage increasing trends towards water scarcity and drought exposure in the river basin districts. Arguably, one of the reasons for the lack of appropriate management practices is the relative failure to implement clear and efficient public policy responses to reduce scarcity and drought vulnerability in the area. The failure of public policy is often associated with the lack of clear legislative rules and analysis on the expected economic impacts associated with baseline conditions. In this respect, Outputs 3 and 4 of EPI water aim to support the development of the Tagus-Segura case study by:

1. Illustrating the key WS&D European policy tools and recommended actions that would frame the development of policy choices at the Member State level. The current legislative framework and actions to target scarcity and droughts are reviewed in this report in an attempt to identify the links between the existing policy context and the proposed EPIs (drought insurance, water trading and smart pricing).

2. Investigating the current status of economic drought impact evaluations in Europe and elsewhere. The second part of this report identifies and discussed state of the art in the development of strategies and consistent methodologies to quantify drought impacts and the economic losses from drought. This is done under the rationale that better quantification of drought impacts will also improve understanding in the case study of the possible benefits resulting from the implementation of the proposed EPIs (drought insurance, water trading and smart pricing).
2. Water Scarcity and Drought – Outline of policy tools and recommended actions at the EU and MS level.

2.1 Introduction

The increased frequency and augmented impact of water scarcity and drought events registered across Europe in the past three decades has called for greater attention from EU officials and institutions on these issues. Within this timeframe, drought occurrence has not only increased in Southern EU countries like Portugal, Spain and Greece, but areas at risk have expanded to include river basins in Northern countries like the UK, Denmark and Sweden (Kossida et al., 2012). By 2007, at least 11% of the European population and 17% of its territory had been affected by water scarcity (COM, 2007), and recent trends predict that half of the EU river basins will be affected by 2030 (COM, 2012a). This sets the stage for increased vulnerability of aquatic ecosystems to extreme weather events (COM, 2012a). Furthermore, external forces like climate change and population growth exacerbate the pressures originating from water over-use, illegal abstraction and inadequate allocation, among other drivers.

The emergence of policy tools and recommendations to cope with water scarcity and drought (WS&D) issues within recent European strategies is a manifest recognition of such problems and their drivers at the national and supranational level. In addition, there has been expressed interest in establishing channels of cooperation across the different government levels to facilitate compliance with such policy mechanisms (COM, 2012b). Nevertheless, the efforts made so far to halt the impending water supply shortage that threatens certain areas of the EU have been only partially successful (COM, 2012a).

The following sections present a brief overview of the context surrounding European WS&D policy and an attempt to elicit the links between this policy context and the proposed EPIs for the WP4 Tagus-Segura case study in EPI-water (drought insurance, water trading and smart pricing) through a review of the policy tools and recommended actions set out in the past years by European and Member State (MS) authorities.

2.2 Overview of the policy context

Water Framework Directive

While water quantity issues are not its main concern, the WFD includes provisions that suggest a balanced relation of abstraction and recharge of water and contemplates measures that have direct effects on water availability (Bayer et al., 2013). Furthermore, as stated in the blueprint document, the Commission will
encourage the uptake of those requirements of the WFD which are related to drought risk management. This is already being done, for instance, through recommendations on the first round of RBMPs which promote a more conscientious integration of drought risk management issues (COM, 2012a). Especially important is to exploit the potential of economic policy instruments implemented under Article 9 of the directive. Such instruments are expected to lead to greater efficiency in water management and allocation and will be discussed in more detail in subsequent sections of this chapter.

Communication on Water Scarcity and Droughts in the European Union

In 2007, the European Commission published a Communication on Water Scarcity and Droughts in the European Union, proposing that European institutions focus on seven main policy options to tackle WS&D, namely (COM, 2012c):

- Putting the right price tag on water
- Allocating water and water-related funding more efficiently
- Improving drought risk management
- Considering additional water supply infrastructures
- Fostering water efficient technologies and practices
- Fostering the emergence of a water-saving culture in Europe
- Improve knowledge and data collection

Apart from identifying the main challenges and outlining policy options and recommended actions to address them, the communication included a series of good practice examples which helped to frame and support the discussion. Unfortunately, the measures developed and implemented by MS in reaction to the communication were found to be limited and in some cases even contradictory to the achievement its objectives (Strosser et al., 2012). This confirmed the need for a new impulse from the EU to step up efforts towards the protection of water resources in the region (see paragraph below on the Blueprint to safeguard Europe’s Water Resources).

Reform of the EU Common Agricultural Policy

The communication described above and many other policy documents, including rural development plans at the MS level, acknowledge the agricultural sector as being a principal actor which can both strongly influence and be affected by WS&D issues. While initially the environmental concerns of the Common Agricultural Policy (CAP) centered mainly on biodiversity and water pollution (Lowe and Baldock, 2000), this focus has widened in the latest years to include water scarcity, flood and drought risk as well. Furthermore, proposals for the reform of the CAP which contemplated the integration of WFD requirements as part of the cross-compliance regulations have been recently agreed by the European Commission, the Council and the Parliament. Such an adaptation is expected to be highly influential in integrating EU policy horizontally and in strengthening the WFD.
Report on the review of the European Water Scarcity and Droughts Policy

While the communication on water scarcity and drought of 2007 was a strategic step stone to streamline efforts, the actual changes in policy driven by it still remain limited. This was one of the findings of the Report on the review of the European Water Scarcity and Droughts Policy, published in late 2012 to evaluate the status of WS&D policy and the advances made on the seven points proposed in the aforementioned communication. Here, the Commission stated, “limited progress has been achieved in implementing the policy instruments identified in the 2007 Communication” (COM, 2012a). Apart from the detailed review of the progress made on each of the policy options, the report includes a section with the results of a screening exercise describing how WS&D issues were addressed in the RBMPs submitted by the MS.

Blueprint to safeguard Europe’s Water Resources

Finally, the Blueprint to safeguard Europe’s Water Resources was published in late 2012 and was developed on the basis of a series of reports commissioned by the European Commission, including the analysis of the first RBMPs, the review of the WS&D policy and the fitness check of European freshwater policy, among others. This strategy document outlines the need for integration across policy sectors and for recognising the natural environment as a stakeholder, and proposes to address these needs by adopting a policy mix that includes approaches like payments for ecosystem services, incentive pricing and ecological flow schemes. The document sets the long-term objective of ensuring the sustainability of economic activities which influence the quality and quantity of water resources while also being dependent in the availability of them (COM, 2012a).

EU Strategy on adaptation to climate change

Published in April 2013, the strategy on adaptation is aimed at encouraging the development and implementation of adaptation action by the MS, with special focus on win-win, low-cost and no-regret options. Under such category of options, the document includes sustainable water management. While the term scarcity is not found in the communication text, several references are made to drought and to the mounting frequency of heat waves and forest fires, an impact of climate change with close links to water demand and availability. As one of the actions to be promoted under its adaptation strategy, the Commission included the expansion of insurance and other financial products in the context of natural and man-made disasters. This is expected to enhance the resilience of the European economy in the face of climate change.

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1 Ecological Flow Schemes are the flow regimes required by aquatic ecosystems to maintain their essential processes in operation and ensure good ecological status (COM, 2012b).
change. Lastly, the communication also touches upon the importance of financing adaptation action and notes the availability of funding streams at national and EU level that support drought management implementations (COM, 2013).

2.3 Links between the current WS&D policy and the proposed EPIs

The use of economic policy instruments (EPIs) to aid in the management of water resources has been explored in different countries throughout the years. While the outcomes and experiences resulting from these explorations have been varied, the new challenges posed by intensified climate conditions and increasing demand of recent years are prompting policy makers to fine-tune existing mechanisms and develop innovative ones. In the European water management context, economic policy mechanisms like water trading, smart pricing and drought insurance have been commonly referenced in strategy documents for their potential to aid in the correction of issues like over-allocation, inefficient use and inadequate risk management, respectively. However, recent studies like the Gap Analysis of the WSD Policy in the EU have questioned the level of ambition and attention placed by MS on addressing WS&D issues and on integrating them as factors into their choice of policies and actual implementations (Strosser et al., 2012). The Commission Staff Working Document on the implementation of the WFD also lists a series of shortcomings in the reported plans and actions of MS to deal with WS&D issues. Furthermore, the document points out the limited extent to which incentive pricing policies and economic instruments have been implemented in MS so far (COM, 2012d). The following paragraphs discuss on some of the references made to these instruments and cases of actual implementation in the EU.

Water Trading to correct over-allocation of water resources

Building a policy framework that ensures the efficient allocation of water resources among multiple users while upholding equity and economic sustainability can be a complex endeavour. This is especially true in regions like the Tagus and Segura river basins in Spain, drought prone areas that exhibit an increasing tendency of water scarcity in the last years. Evidence has shown that under specific conditions (e.g., well-defined and enforceable water rights, low transaction costs, transparency) the establishment of water markets can result in benefits to the multiple stakeholders directly and indirectly involved. Water markets can help reveal the value of water, promote competition and balance demand (OFWAT, 2010; Cave, 2009; Bate, 2005). Water markets have been operational in regions outside Europe since the past century (e.g., Australia, Chile, United States). The water market in the Murray-Darling Basin of Australia is a common reference for countries interested in implementing water trading. The initiative started in the 1980s and it is regarded as a major success story based on its capacity to drive innovation and efficiency and to increase the resilience of the sector during drought and water shortage periods (Young, 2012). The Chilean case is one where free market ideology, localized trade and a strong institutional framework converged to yield a market structure which
was successful in allocating water resources to the most valuable uses, resulting in substantial economic gains (Meinzen-Dick, 2007). In the U.S., the success of the water trade of the Colorado–Big Thomson Project is authenticated by its high participation levels, which in turn have been attributed to the transparent and straightforward character of the system (OFWAT, 2010). Within the EU, only Spain has had considerable experience with water trading schemes. In its initial form, the Siurana-Riudecanyes water permit trading scheme was set up by an association of irrigators to secure water supply to the city of Reus and the agriculture of the region. As early as 1911, water titles were being allocated through fixed price public offering on the basis of financial contributions made by members of the association to fund the required hydraulic infrastructure for the area. Since then, water titles can be traded between members of the association (i.e., farmers and municipal water supply companies) (Tarrech et al., 1999). Informal water trading has existed in the country for centuries, and over the years the Spanish experience has evolved into a more versatile market structure where trading can also be done formally and which has already proven its potential operating under drought conditions between 2005 and 2008 (COM, 2012b). In several of the policy documents described in the last section, the European Commission acknowledges the need to correct over-allocation of water resources and makes direct reference to the potential of water trading. As stated in the blueprint document, setting up a “sustainable overall cap for water use” and establishing market structures at economically viable scales (i.e., keeping administrative costs at reasonable levels) could facilitate achieving more efficient allocation and reduced water stress (COM, 2012a). According to the report on the review of WS&D policy, this could also set the stage for payment for ecosystem services (PES) schemes, promoting modes of resource allocation that integrate provisions for economic and environmental sustainability (COM, 2012b). Additionally, in providing a more accurate outline of the availability of water resources at different geographical scales, the work on water accounts being developed in collaboration with the EEA will facilitate the establishment and administration of water trading schemes (COM, 2012a). Finally, the adaptation or construction of such trading schemes should be done under the premise of ecological flows, contemplating the allocation of water resources to water-dependent ecosystems and allowing for continuity in their function and provision of ecosystem services (COM, 2012b). In many cases this will require the consideration of new infrastructure and information systems, as well special governance and regulation mechanisms to ensure availability, equity and transparency (Strosser et al., 2012).

Box 1 - The prospects for water trading in England and Wales

The lack of a thorough restructuring of the water and wastewater sector in England and Wales prior to its privatization in 1989 resulted in notably low upstream competition levels (Stern, 2010). Today, this adds up to the multiple challenges to water resource management posed by changing climatic conditions, growing population and a strained economy. While water trading has been a possibility for
some time in both countries, levels of participation in the water market have been continually low (Synovate UK, 2008). In spite of the investment and improvements made since privatization, the hydraulic infrastructure necessary to promote higher trading activity is still missing (Möller-Gulland, 2010). Such a setting has triggered an ongoing discussion on the need of a reformed water management framework capable of increasing competition, innovation and efficiency. In exploring the options for reform, incumbent authorities like the Environmental Agency (EA) and OFWAT (the water services regulator in England and Wales) have assessed and consulted on the adequacy of activating market mechanisms, particularly water trading. This has driven substantial efforts to scope the current context, outline the potential benefits and identify the barriers to successful development of upstream water markets (Cave, 2009; Möller-Gulland, 2010; Stern, 2010; Young, 2012). A good proportion of the resulting literature has recognized the potential of water trading inter alia to (Stern, 2010; OFWAT, 2010; Möller-Gulland, 2010):

- maximize efficiency by revealing the relative value of water and by allocating resources to the most productive uses
- reduce the costs of meeting demand by granting access to market to new players and separating networks from services
- ease water stress and secure supply by incentivizing water intensive industries to locate their facilities away from water-scarce areas
- provide commercial incentives for investment in the infrastructure necessary for increased water trading

While a final decision on the way forward has not yet been met, the current state of the discussion seems to point towards the transition to a simplified, streamlined water supply licensing process and a more transparent and visible water market for England and Wales.

**Smart Pricing**

Pricing systems are a main element of EU water policy aimed at the protection and efficient use of water resources. The EU’s blueprint document regards water pricing as “a powerful awareness-raising tool for consumers and combines environmental with economic benefits, while stimulating innovation” (COM, 2012a). By including environmental and resource costs (ERC) into water service tariffs, the WFD requirement of cost-recovery points towards establishing a pricing system that deals better with environmental externalities. This is also acknowledged in the report on the review of WS&D policy, which calls for the set up of tariffs where they are lacking and the widening of charges and taxes to promote sustainable abstraction.
Besides ensuring the sustainability of water service provision, metering and volumetric pricing also have a positive effect on water consumption. For instance, a study by the OECD showed that a 20% decline in domestic water consumption was registered in the six summer months after the introduction of metering in the UK (OECD, 2009). In comparing volumetric rates and flat rates, Hernandez and Llamas showed that the former resulted in 25-35% less water used (Hernandez and Llamas, 2001). Similarly, agricultural areas under binomial tariff schemes in the Guadalquivir river basin in Spain exhibited consumption levels 10-20% lower than those under flat rate schemes, regardless of the flat rate level (Rodríguez-Díaz, 2004 in EEA, 2009). While it is clear that a wide range of local and regional factors (current legal framework, water availability, price elasticity of demand, capacity for water saving) influence the actual effect of water pricing schemes (Bogaert et al., 2012), the findings above suggest their potential to reduce consumption levels. Furthermore, pricing is an aspect of water policy that is of main concern to a wide range of economic sectors and policy areas, and thus it can be used as one of the threads to drive innovation and horizontal integration across multiple sectors. For instance, in seeking to incorporate the ERC requirement to rural development policy, the Commission has proposed water pricing and cost-recovery to be a pre-requisite to the allocation of funds from such plans (COM, 2012a). Similar measures have been already adopted in France, where eligibility for CAP subsidies is conditioned to metering of abstracted water (ACTeon, 2011). Regarding innovation, the efforts made under the European Innovation Partnership (EIP) on Water will also be crosscutting and will aim for integrated solutions that can address water scarcity issues in urban and rural areas. In this line, to also include provisions of WS&D policy, the pricing mechanism proposed in the WFD could be expanded into a smart pricing system that factors issues like current climatic conditions, geographic location and productivity of water use types into the price of water services. As discussed in the aforementioned gap analysis, there is a need within current water policy to distinguish between water productivity levels and to promote a shift of water-intensive activities towards areas whose climatic and geographic conditions are better suited to sustain their water needs. Reflecting the variances in such factors through differentiated charges can aid in achieving these goals and lead to water balance across river basins (Stroesser et al., 2012). While these ideas are gaining higher clout at the EU level, recent studies including the assessment of the RBMPs have shown that even support initiatives and implementations such as metering, which should set the groundwork for operational incentive pricing and cost recovery, are still quite limited at the MS level (COM, 2012e; COM, 2012d). Currently, there is still a significant gap between the structures of pricing schemes in the different MS and a smart pricing system as proposed. While conditions, institutional and legal frameworks vary widely across the EU, a water-pricing scheme which was commonly found by a study scoping several European MS is a hybrid model that combines fixed and variable components (service charges and volumetric rates). However, the lack of metering infrastructure is often responsible for the prevalence of flat rates (De Paoli et al., 2013). Developing the pricing schemes to expand them from their present state to one that accounts for
the issues discussed above will require the implementation of support actions that allow for measuring and control. For example, the construction of demand forecast models, like are currently being developed in Germany, will result in new capacities based on information structures that can be useful in establishing more dynamic pricing mechanisms.

Box 2- The potential of the water pricing framework in Portugal

Smart water pricing schemes are still generally in the discussion phase, and while the mainstreaming of its more technical aspects (i.e., full rollout of smart metering systems) is being pursued in Italy, Malta and Greece (De Paoli et al., 2013), a sufficiently developed case to portray the policy aspects of smart pricing is lacking. The actual impact of these technical implementations might be questionable without an accompanying pricing policy. Nevertheless, some countries already count with pricing frameworks that allow for flexibility and adaptability, thus providing the groundwork for smart pricing. This is true for systems that consider the specific conditions of the region in which they are embedded and leave space to integrate the distinct characteristics of the sector to which they are applied. The case of Portugal, for instance, outlines a pricing policy design that pursues basic principles like full-cost recovery, affordability and transparency by incorporating these into national law. At this same level, it makes provisions for the possibility of seasonal variable water tariffs (peak-load pricing). This makes up a basic framework from which key EU requirements and national interests can be safeguarded. To complement this overarching framework while leaving room for regional and sectoral specificity, local authorities are granted competences to approve water service tariffs and define social price plans for poor households, amongst others. To ensure the compatibility between the national and local decisions, the country’s Water and Waste Services Regulation Authority (ERSAR) plays an important linking role and works closely with municipalities (Pires, J.S., 2007 in OECD, 2010). In the specific case of tariffs, ERSAR advocates for “adequate return of investment and services operational costs, as well as of environment and scarcity costs”. Such a water pricing policy structure has the capacity to accommodate new pressures arising from external drivers, and thus could be further developed in the coming years to internalize the costs of solutions that may become ever-more necessary under drought and scarcity conditions (e.g., non-conventional water sources).

Drought Insurance

Adverse weather conditions (such as frost, hail, thunderstorms or droughts) can cost a farmer a season’s crop. Agricultural insurance is seen as an important instrument

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to help farmers manage risks associated with production. Given the uncertainties concerning the incidence of droughts; residual production and income risks would still persist despite the application of water markets or smart pricing, as examples of some of the policy tools described previously (Volaro et al., 2013). Ultimately, without compensation mechanisms that cover total income losses faced by farmers due to adverse weather conditions, the real insurance system for income stabilization is often found in alternative water resources. Illegal water abstractions, especially from aquifers (as an example in Spain), offer a source of income coverage from the production losses that are not compensated for during a period of droughts. As a criticism to the current regulatory framework in Spain these sources of water are scarcely controlled and rarely pursued and punished (Gómez and Pérez, 2012).

In this context, drought insurance systems can be used as a tool to introduce the necessary incentives to reduce water overexploitation during drought events and the high costs of the compensation subsidised by the government (Pérez and Gómez, 2012). At the same time, this ties well with action 8 of the aforementioned EU Strategy on adaptation to climate change. This action proposed by the Commission consists in promoting insurance and other financial instruments to enhance the resilience of investment and business decisions in the face of natural and man-made disasters (COM, 2013). Two draft EU regulations tendered under the current revisions of the CAP contain guidelines for the development of agricultural insurance schemes in relation (to some extent) with extreme weather events:

1. The Commission proposal for a regulation on support for rural development by the European Agricultural Fund for Rural Development (EAFRD)
2. The Commission proposal for a regulation establishing a common organization of the markets in agricultural products (Single CMO Regulation)

The first draft regulation supports farmers to cover the premiums they pay for crop, animal and plant insurance, as well as the setting up of mutual funds and the compensation paid by such funds to farmers for losses suffered due to adverse climatic events and animal or plant diseases or pest infestation. The Commission also suggests an income stabilisation tool in the form of a mutual fund to support farmers facing a severe drop in their income. Careful estimation of farmers’ income levels must be assured in order to avoid overcompensation from the coupling of this tool with other support instruments and/or private insurance schemes (COM, 2011a).

The second draft regulation makes reference to harvest insurance and crisis prevention and management (i.e., avoiding and dealing with crises) on the fruit and vegetable markets. Here, the Commission suggests supporting producers to cover the premium of insurance against losses from natural disasters, adverse climate events, diseases or pest infestations. Furthermore, the repayment of capital and
interest on loans taken by producer organizations to finance crisis prevention and management measures is also proposed as eligible for assistance (COM, 2011b).

Table 1. Proposed support rates for losses from natural disasters, adverse climatic events, diseases or pest infestation

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<td>Proposal for regulation on Single CMO</td>
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</tbody>
</table>

* These amounts may be increased in exceptional cases taking into account specific circumstances to be justified in the rural development programmes.

Box 3 - The agricultural insurance market in Spain

The legal framework which regulates the agricultural insurance system in Spain is laid out in the 87/1978 Law of December 28, on combined agricultural insurances, and the 2329/1979 Royal Decree of September 14, which approves the Regulation for the application of the Law. For the practical implementation of these regulations, the Government establishes the “Agricultural Insurances Annual Plan”, which defines the rules to be held into account in the application of the agricultural insurance schemes in the country.

In Spain, all companies involved in the agricultural insurance market operate within a pool, which assumes the risk in a coinsurance regime. This means free competition exists only in the retail sector. The insurance company “pool” is managed by the “Spanish Pool of Insurance companies for the Combined Agricultural Insurances, S.A.” (Agroseguro, S.A.4). Agroseguro’s responsibilities include the elaboration of insurance contract models and the fixing of premiums for each insurance product. Such proposed premiums must be justified (with economic data and actuarial results) to the sector regulator, the Dirección General de Seguros y Fondos de Pensiones - ENESA5 (entity depending from the Economy Ministry). ENESA manages the control mechanisms and regulates the sector in close liaison with farmer’s unions.

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3 It is important to note that discussions between EU Member States, the European Commission and the European Parliament on these proposals related with the latest reform to the CAP, are ongoing. The draft regulations are subject to change at any time, up to their adoption by the European Council and the European Parliament, which is currently expected during 2013.

4 http://www.agroseguro.es/

5 http://www.ensen.es/
As of 2006, a total of 33 national as well as foreign private insurance companies were registered in the coinsurance pool. Competition is ensured by impeding control of the market by a single company. MAPFRE is the leader, with a 22.5% of the total market share. The participation of farmers in the system is voluntary and there are subsidies in place from the public administrations. It is said in official documentation that the average subsidy is about 50% of the total insurance premium. From this percentage, 40% comes from ENESA and the rest (10%) comes from the governments of the Spanish Autonomous regions.

In 2012, 490,000 agricultural insurance policies were issued in Spain covering around 400,000 farmers. A total of 795 Million Euro were claimed by farmers to cover economic losses originated mainly due to adverse weather events, including droughts. This represents an increase in claims of around 65% in comparison with 2011. In relation with damaged surface area, new figures released by Agroseguro highlight that in 2012, a total of 1,630,527 hectares were reported compared to a total of 863,000 hectares for the period 2007-2011 (Agroseguro, 2013).

According to proposed recent EU law and for the design of new insurance schemes, it is important to note the relationship between subsidies and farmers’ insurance premium and how it compares to application in other countries. A recent review by the World Bank on the public cost of agricultural insurance subsidies highlights that Spain is among those countries (Italy and the USA) where public support to agricultural insurance represents more than twice the premium paid by farmers. Furthermore, the report illustrates that in Spain, total subsidies as percentage of 2007 producer premium were around 300%. In contrast, France is in the lower end of the spectrum with a contribution through subsidies close to 20% of the premium.

The legislative framework developed in Spain since the passing of Law 87/1978 has made it possible to devise and implement an agricultural insurance system that has become one of the most effective instruments available to farmers to help safeguard their revenue from the consequences of drought and other risks that they cannot control. This is recognized by experts in the field and by the farm sector itself (Burgaz, 2008). The model is well established and valued by farmers. In this respect, there is a high possibility that agricultural insurance has the best chances of forming an effective policy instrument for the management of drought risk. Although, some future reforms of the agricultural insurance schemes in Spain may need to adjust to proposed new EU regulations, in where Member States may be asked to limit the amount of the premium that is eligible for support through the EAFRD by applying a maximum ceiling of 65%.
3. Costs of drought damages

Climate and weather influence a large proportion of economic activities. In developed economies such as the U.S., the impacts can amount to about 25 percent of the GDP (NOAA 2002). Therefore, the consequences of severe weather events can have serious economic impacts. As an example, in 2012 alone, weather-related natural catastrophes and the summer-long drought, which affected the Midwest and surrounding states in the US, caused extensive economic damage, especially to the agricultural sector. According to Munich Re, crop losses totaled $20 billion in 2012, making it the largest loss in U.S. agricultural history. Approximately $15 billion to $17 billion of these losses are covered by the federally subsidized multi-peril crop insurance program (Pavkow, 2013).

Specifically in relation to droughts, the European Commission (2006) estimated the costs of droughts in Europe over the last 30 years to be at least €85 billion. The overview table 2 highlights some of the total cost estimates found in the literature specifically related with the total costs of drought events.

![Table 2 Examples of total cost estimates of economic damages related to droughts](image)

Although estimates vary, the take home message from table 2 is that the economic impact of extreme weather events, including droughts, can reach considerable amounts. As a matter of fact, overall cost estimates are increasing as a consequence of the more pronounced impacts due to climate change. This in addition to the fact that more accurate estimates are coming through as there is an increasing effort made in understanding the estimation of economic impacts due to a policy demand for better informed policy advice in the possible
consequences. As a result, more and more detailed economic assessments of drought impacts are slowly becoming available in the grey and academic literature.

As an example, the drought of 1991-1995 in Spain\(^6\) had the following documented impacts:

- **Agriculture:** an observed annual decline in production of around 1.2-1.9 Billion Euro between 1992-1995 compared to earlier and later years. This resulted in a 60\% increase in the uptake of farm insurance policies for rain-fed irrigation in 1999 compared to those recruited in 1995 (building to a total insured capital of 4.6 Billion Euro);
- **Urban Water supply:** 12 million people suffered restrictions in 1995 (acute problems in the cities of Seville, Cadiz and Palma de Mallorca)
- **Environmental effects:** not quantified (mortality of fish fauna in ponds and wetland birds was seen)

The estimation of the economic benefits resulting from the implementation of the three proposed EPIs in the Tagus-Segura case study mainly resides in the cost savings that could be derived from efficient adaptive management to scarcity and droughts in the river basins. This mainly represents a reduction in scale of the possible impacts that could come as a result of a drought period (as outlined in the example above, the amount of these savings could be considerable). Fundamentally, the economic analysis of drought damages provides crucial information for decision support and policy development in natural hazard management and planning for adaptation to climate change (Logar and van der Bergh, 2012).

At this moment, the existing literature on the types of economic costs from drought events and methods for their estimation is scarce, fragmented and heterogeneous (Markandya et al. 2010; Logar and van der Bergh, 2012). This chapter aims to review, explore and understand the economic impacts of water scarcity and droughts with a focus on identifying existing cost analyses, values and methodologies with the underlying aim of improving both data and knowledge gaps on the economic impacts of droughts that is relevant for the Tagus-Segura case study.

The remainder of the chapter is organized as follows: Section 1 reviews the fundamental economic principles of water resource depletion and degradation. This section includes a brief overview of different valuation methods used in the literature and an introduction of the different concepts of economic value that are relevant for drought impacts assessments. Section 2 follows with a literature review on cost assessments of drought damages and drought mitigation and adaptation policies with a focus on understanding the main observed economic impacts of droughts, explore relevant types of costs and discus their relevance, associated methodological challenges for their estimation and potential use to inform adaptive management.

\(^6\) [http://hispagua.cedex.es/sites/default/files/especiales/sequia/efectos_sequia.htm](http://hispagua.cedex.es/sites/default/files/especiales/sequia/efectos_sequia.htm)
3.1 The economics of water resource depletion and degradation: a conceptual framework

Water itself has a value; this is most notable for drinking, irrigation, food production, sanitation, energy use, forestry, tourism, housing etc. Indeed, for some activities, it is a commercially supplied product (e.g. the IT and medical sectors require high purity waters). It is fundamental for society and for the economy and underpins most of our activities.

The lack of water can have significant effects on health, livelihoods, the economy, and on the operations and efficiency of industry across most sectors. All sectors of the economy depend on water directly and/or indirectly. The agricultural sector depends on water for crop and livestock production; the energy sector for hydropower and for cooling at thermoelectric power plants; the tourism sector for the natural beauty provided by rivers, lakes and the sea. Where water is scarce, water security concerns can arise between users or between regions (e.g. in trans-boundary contexts).

While it is well accepted that water has an economic value, the real problem for its estimation is that unfortunately, very often water cannot be treated as another common trading commodity due to its special characteristics, and therefore, its economic value should not be directly equated with its market price. Several authors (Morris;2004, Hanemann, 2006 among others) have identified the main reasons for this special treatment:

1. water is a fugitive, re-usable resource which is difficult to control and account for;
2. water is often a common property, with open access and ill defined property rights;
3. it provides public goods, such as the public health benefits of clean water;
4. water is used in many different ways which often result in external consequences to other water users and the environment (externalities);
5. it is essential for life, without close substitute: a need rather than a want
6. water is subject to uncertain supply associated with climate variations;
7. water has significant economies of scale associated with its managed supply; and,
8. water is an integral part of the functioning of ecosystems.

In theory, a sustainable use of water could be achieved when the full costs of supply equal its full price (value). The extraordinary characteristics of water as a resource make it difficult to design markets for its right allocation (through entitlements for example) amongst competing users and for the correction of externalities. Fundamentally, the resource has many different uses which can be assigned many different values.

**Total Economic Value**

The overall aim of this chapter is to understand the economic impact of droughts through an analysis of the literature. By doing this, the exercise aims to shed some light onto how to find out the total economic value of scarce water resources, which for the Tagus-Segura case study consists of an improvement in water quality and quantity related to the definition of “Good Status” according to the EU WFD. The total economic value is measured from people’s preferences for the use of the resource or by measuring the different levels of utility that people would place on them. The value for the entire population affected is established by an exchange transaction reflected in the sum of each person’s value for environmental
improvements or in other words, the area under the demand curve of the environmental good that is improved.

Figure 1 Pathways for the estimation of total economic value.

Fundamentally, TEV recognises the distinction between the value that individuals derive from using the environmental resources, i.e. use values, and the value that individuals derive from the environmental resource even if they themselves do not use it, i.e. non-use values (Birol et al, 2006).

Use values can be:

1. Direct (DUV), such as when an individual makes actual use of the environmental asset improved, for example, fishing where it wasn’t possible to catch a fish before the improvements in water quality took place;
2. Indirect use values (IUV), such as the benefits derived from ecosystem functions gained, for example, where recreational activities are created or enhanced due to water quality improvements, individuals can benefit in the form of increased recreational opportunities and finally;
3. Bequest or option values (BV/OV) which measure individuals’ preferences to ensure that their heirs or future generations will be, for example, able to enjoy the improved resource in the future (make use of it).

Non use values are often called existence values (XV), defined as the economic value placed by people for improvements to the quality of a river due to some moral and/or altruistic reasons, or for the mere pleasure of knowing that the river’s water has been enhanced.

Furthermore, Willingness To Pay (WTP) and Willingness To Accept (WTA) are the two standard measures of economic value. WTP is the appropriate measure in the situation where an agent wants to acquire a good. Minimum willingness to accept (WTA) compensation is the appropriate measure in a situation where an agent is being asked to voluntarily give up a good. Both of these measures are Hicksian consumer surplus measures and are often defined net of the price actually paid or received (Carson, 2000). Whether WTP or WTA is the correct measure depends on the property right to the good. If the consumer does not currently have the environmental good and does not have a legal entitlement to it, the correct property right is WTP. If the consumer has a legal entitlement to it and is being asked to give up that entitlement, the correct property right is WTA.

Valuation Methods

Table 3 outlines the different valuation methods used in the environmental economics literature for the estimation of the different components of TEV. This table classifies methods according to their capacity of using conventional, surrogate or hypothetical markets for the estimation of use and non-use values and also, on their ability to estimate WTP and/or WTA. The Contingent Valuation Method (CVM) and Conjoint Analysis (CA) can derive monetary values from both WTP and WTA and use/non-use values. All methods based on revealed preferences can only estimate use values from WTP (i.e. travel cost method (TCM), Hedonic Pricing applied to the property market (HPPM), Averting Behaviour (AB) and market prices), with the exception of Hedonic Pricing applied to Labour Market (HPLM), which can only derive WTP use values. Finally, the results of all these methods can be used for benefits transfer between sites. It is not the scope of this paper to introduce in detail the practicalities surrounding the application of each valuation method7.

7 For a detailed review of these methodologies; see for example: Garrod and Willis (1999), Smith (2004), And Bateman et al. (2002) for CVM; Markandya et al., (2002); or Parsons (2003) for TCM; Melichar and Ščasný (2004) or Viscusi, (1993) for HPPM.
### Table 3 Component of TEV of water resources and appropriate economic valuation methods

<table>
<thead>
<tr>
<th>TEV component</th>
<th>Economic valuation methods a</th>
</tr>
</thead>
<tbody>
<tr>
<td>Direct use values</td>
<td></td>
</tr>
<tr>
<td>Irrigation for agriculture</td>
<td>PF, NFI, RC, MP</td>
</tr>
<tr>
<td>Domestic and industrial water supply</td>
<td>PF, NFI, RC, MP</td>
</tr>
<tr>
<td>Energy resources (hydro-electric,</td>
<td>MP</td>
</tr>
<tr>
<td>fuelwood, peat)</td>
<td></td>
</tr>
<tr>
<td>Transport and navigation</td>
<td>MP</td>
</tr>
<tr>
<td>Recreation/amenity</td>
<td>HP, TC, CVM, CEM</td>
</tr>
<tr>
<td>Wildlife harvesting</td>
<td>MP</td>
</tr>
<tr>
<td>Indirect use values</td>
<td></td>
</tr>
<tr>
<td>Nutrient retention</td>
<td>RC, COI</td>
</tr>
<tr>
<td>Pollution abatement</td>
<td>RC, COI</td>
</tr>
<tr>
<td>Flood control and protection</td>
<td>RC, MP</td>
</tr>
<tr>
<td>Storm protection</td>
<td>RC, PF</td>
</tr>
<tr>
<td>External eco-system support</td>
<td>RC, PF</td>
</tr>
<tr>
<td>Micro-climatic stabilisation</td>
<td>PF</td>
</tr>
<tr>
<td>Reduced global warming</td>
<td>RC</td>
</tr>
<tr>
<td>Shoreline stabilisation</td>
<td>RC</td>
</tr>
<tr>
<td>Soil erosion control</td>
<td>PF, RC</td>
</tr>
<tr>
<td>Option values</td>
<td>CVM, CEM</td>
</tr>
<tr>
<td>Potential future uses of direct and indirect uses</td>
<td>CVM, CEM</td>
</tr>
<tr>
<td>Future value of information of biodiversity</td>
<td>CVM, CEM</td>
</tr>
<tr>
<td>Non-use values</td>
<td></td>
</tr>
<tr>
<td>Biodiversity</td>
<td>CVM, CEM</td>
</tr>
<tr>
<td>Cultural heritage</td>
<td>CVM, CEM</td>
</tr>
<tr>
<td>Bequest, existence and altruistic values</td>
<td>CVM, CEM</td>
</tr>
</tbody>
</table>

With modifications adopted from Barbier (1991), Barbier et al. (1997), Woodward and Wui (2001), Brouwer et al. (2003), and Brander et al. (2006).

a  Acronyms refer to production function (PF), net factor income (NFI), replacement cost (RC), market prices (MP), cost-of-illness (COI), travel cost method (TCM), hedonic pricing method (HP), contingent valuation method (CVM), and choice experiment method (CEM).
3.2 Literature review on cost assessments of drought damages and drought mitigation and adaptation policies

Summary of the main observed effects of droughts

Before looking into detail at the definitions of the cost typologies and associated methodological challenges for their estimation, it is important to map out the most observed general effects of droughts to different economic sectors. The following effects have mainly been taken from a review by Kraemer (2007).

The direct impact of droughts on agriculture and breeding of animals can come mainly from higher costs and from reduced productivity, both immediately and over the longer term. Immediately, as agricultural plant productivity is generally reduced, the size, the total weight and the general health of animal herds is affected. Where the lack of rainfall can be compensated by irrigation from alternative water resources, droughts induce additional costs for water abstraction and pumping. Such measures can indirectly have a negative impact on natural ecosystems. In addition, the effects linger on after a drought has ended, especially where the drought, wind or subsequent precipitation falling onto barren soil has caused erosion, loss of organic soil matter, or degradation in soil structure. Drought-induced stress physiologically also affects the health of perennial plants and makes them more vulnerable to pests and disease in subsequent years. Animal herds may need additional feeding during and after droughts, when fodder may have to be bought in at higher prices than usual. Where herd sizes need to be reduced during drought periods, revenue from animal or meat sales are usually significantly lower than normal, and it can take some time to rebuild herds to original levels.

The impact of droughts on the energy sector can be summarised as:1) increased power demand for air-conditioning and other cooling processes (and water pumping); 2) Reduced water availability for hydropower, which may then need to be compensated by increased use of fossil fuels and an indirect increase in energy prices; 3) reduced water flow or warmer water flow in rivers, reducing the capacity for cooling thermal power plants (nuclear and coal). The economic impact of droughts on the energy sector differs greatly depending on the electricity generation mix, the capacity of water bodies and so on. Where power generation has to be lowered in areas highly dependent on hydropower or where large power plants are located on inland rivers with exceptionally low flow, the economic consequences for electricity using sectors can be significant.

In the water supply sector, the effects of droughts involve interruptions to supply and further restrictions on water use (bans on the use of hosepipes, lawn watering, other). In addition to demand reduction (which tends to result in rising volumetric prices for water) and an increased competition for water rights (which become more scarce and expensive during droughts). Finally, additional costs can come from further investment to secure future water supplies in face of extreme weather conditions.

The effects of droughts on the tourism sector vary widely and depend very much on the type and timing of the drought and the region affected. In winter, droughts can have a direct impact on the skiing industry as well as an indirect or delayed impact on summer tourism activities, such as canoeing or rafting in rivers carrying meltwater in spring or summer.
summer drought in a coastal region or over islands can reduce the water available for holiday homes, hotels, swimming pools, or the irrigation of golf courses. Where a drought leads to subsequent wildfires and the destruction of attractive landscape features, forests or wildlife, the economic impact on the tourism sector in the region can be significant and prolonged. Other tourist destinations might benefit by attracting more tourists.

Finding a cost typology

There are only few recent attempts at bringing together a consistent cost typology in order to understand the economic impacts of drought events (see for example recent efforts made by the Xerochoře¹ and ConHaz² FP7 projects, the world bank…). The proposed classifications are not a 100% compatible with each other. Although the majority distinguish that costs of droughts can be direct or indirect depending on the type of impact and of a market or non-market nature depending on their methodological estimation. Different cost classifications in the literature relate mainly to the different variations in terminology between these 4 combinations. As the issue becomes a matter of choice, we have opted to follow the classification described in Logar and van der Bergh (2013) from the FP7 CONHAZ project. This project classifies drought costs into direct, indirect and non-market (sometimes also called intangible) costs. An overview of the different cost types is provided in Figure 2. In addition to the costs of drought damages, one can distinguish the costs of mitigation and adaptation to droughts. Such costs can also be classified into direct, indirect and intangible cost categories.

Figure 2 Types of drought costs. Source: Logar and van der Bergh. (2013)

<table>
<thead>
<tr>
<th>Direct costs</th>
<th>Indirect costs</th>
<th>Intangible costs</th>
</tr>
</thead>
<tbody>
<tr>
<td>related to biophysical impacts of droughts; include losses in resource-based sectors</td>
<td>losses induced by changes in resource-based sectors on the rest of the economy</td>
<td>non-market costs associated with environmental and health impacts of droughts</td>
</tr>
</tbody>
</table>

Costs of drought mitigation and adaptation

Direct costs

From an economic perspective, a natural disaster can be defined as a natural event that causes a perturbation to the functioning of the economic system, with a significant negative impact on assets, production factors, output, employment, or consumption. Examples of such natural event are earthquakes, storms, hurricanes, intense precipitations, droughts, heat waves, cold

¹ http://www.feem-project.net/xerochoře/
² http://conhaz.org/
spells, and thunderstorms and lightning (Hallegatte and Przyluski, 2010). In this context, the main direct economic costs from droughts can be summarized as those that reduce the income obtained by land users as a result of the lower productivity of land resulting from reduced availability of water. These 'on-site' costs are experienced either by the land user who degrades the land, or by another user who uses the site subsequently (Low, 2013). In addition, losses may occur in relation with reduced water availability (e.g., water-providing companies, hydroelectric production, water transport), and damages on buildings and infrastructure due to ground subsidence Logar and van den Bergh (2013).

The direct economic impact of droughts on the agricultural sector is the most analyzed cost category in the literature. This is because direct costs are localized as droughts are deeply felt by local agricultural communities or in the regions affected and often production losses are reported through insurance claims.

Evidence from the EU indicates that crop yields fall significantly when production is affected. The reduction in farm income following the 1990–1995 droughts in Spain was estimated at 1.8 billion Euros. The cost of the 2003 drought in France has been estimated at 590 million Euros. In Spain, the production of cereal fell by 42% and wine by 20% in 2005, with payments to the livestock sector added up to 1 billion Euros. Further evidence reports that in France and for the same year, autumn crops fell by 10% and maize by 20% (EC, 2007).

Examples from outside the EU suggest costs in the same order of magnitude. The US Army Corps of Engineers (1994) highlighted that the direct economic losses to California’s irrigated agriculture in 1991 were estimated at only $250 million. This represents less than 2 percent of the state’s total agricultural revenues (California’s $18.3 billion dollar agricultural sector (California Department of Food and Agriculture, 1992). In Australia, direct costs to agriculture and livestock sectors from the 2006/07 drought solely on reduction of farm gross margins were reported to have an overall impact in the Australian economy of around 1% of total GDP. RBA (2006) also reports an impact of around 20% reduction to agriculture’s GDP.

The costs or losses in farm income reported for some EU Member States cannot simply be extrapolated to give an estimate for the European Union as a whole. However, the figures from Europe match those reported from the United States when taking into account the agricultural land surfaces involved (Kraemer, 2007).

Where direct costs to other sectors and other types of costs (including indirect and non-market costs) are reported in detail, their economic impact at the local level often surpasses those reported in agriculture. Martin-Ortega et al. (2009) appraised the total losses of the drought which affected the Spanish region of Catalonia during 2007 and 2008 at a total €1.6 billion and around 0.48 % of the Catalanian total GDP. Direct costs were estimated at a total of 540 million euros per year and represent actual expenses for drought related measures of the River Basin Authority and public water companies and production losses of affected sectors (farming, gardening, swimming-pool companies and hydroelectric production).
Table 4: Summary of direct costs of main economic agents due to the 2007-2008 drought event in Catalonia

<table>
<thead>
<tr>
<th>Sector</th>
<th>Direct costs (M€ per year)</th>
<th>Description</th>
<th>Reliability</th>
<th>% of Catalan GDP</th>
</tr>
</thead>
<tbody>
<tr>
<td>River Basin Authority</td>
<td>77.41</td>
<td>Expenses for drought related measures</td>
<td>High</td>
<td>0.04</td>
</tr>
<tr>
<td>Water suppliers</td>
<td>17.79</td>
<td>Expenses for drought related measures (extrapolation)</td>
<td>Medium</td>
<td>0.01</td>
</tr>
<tr>
<td>Irrigators</td>
<td>62.76</td>
<td>Production losses</td>
<td>Medium to low</td>
<td>0.03</td>
</tr>
<tr>
<td>Gardening and flower companies</td>
<td>210.00</td>
<td>Production losses</td>
<td>Very low</td>
<td>0.10</td>
</tr>
<tr>
<td>Swimming pool and related companies</td>
<td>45.00</td>
<td>Sales losses</td>
<td>Medium</td>
<td>0.02</td>
</tr>
<tr>
<td>Hydroelectric production</td>
<td>127.30</td>
<td>Production losses</td>
<td>Medium</td>
<td>0.06</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>540.26</strong></td>
<td></td>
<td>Medium</td>
<td><strong>0.27</strong></td>
</tr>
</tbody>
</table>


The annual direct costs of the 2007-2008 drought for the irrigation sector in Catalonia were estimated at around 60 Million Euro. In relation with other economic sectors, gardening and flower companies and hydroelectric energy generation were directly hit harder (210 and 130 Million Euro respectively) than agriculture. Agriculture did not suffer the highest losses, and direct losses to the sector represent around 10% of all total direct costs and around 4% of the total estimated losses for the region.

Using the 1991 drought event in the US as an example, whilst direct costs to agriculture accounted for $250 million, costs to hydroelectricity production and Gardening and flowering companies were much larger. CDWR (1991) reported that energy ratepayers in California spent an estimated additional $3.8 billion for electricity during the drought from 1987 to 1992. The direct costs to electric utilities is uncertain but the indirect costs were considerable, as the industry had to replace the lost hydroelectricity with more expensive natural gas and out-of-state power purchases. The replacement costs were mostly passed down to consumers. These costs increased marginal electricity costs to consumers by approximately three cents per kilowatt-hour (CDWR 1991). Based on this estimated marginal cost increase, the drought cost state ratepayers an estimated $21 per person per year (calculated by multiplying estimated lost hydropower production by 3 cents per kilowatt-hour). Another industry significantly affected by the drought was the “Green Industry” (Cowdin and Rich 1994), including landscaping and gardening. Drought-induced direct economic losses to this sector in 1991 were estimated to include the loss of about 5,630 full-time jobs, and a reduction of about $460 million in gross revenue from the 1990 total of $7 billion (Cowdin and Rich 1994).

Furthermore, some groups may actually benefit from the hardships droughts impose on others. For example, drought-induced agricultural losses are likely to increase the prices
received by farmers whose crops are unaffected by a drought. And as seen above, a decline in the production of hydropower increases the demand and the price for alternative sources of energy. When these income transfers are included in the analysis, the aggregate costs of drought to producers tend to decline as the scale of the drought impact assessment is increased. Thus, drought events that are costly at the local level, may be less expensive at the regional level, and negligible at the national level. An analysis of the agricultural economic impacts (measured as the sum of consumer and producer surplus) of drought on California and the nation in 1991 indicated that the national costs of California’s drought were less than 30 percent of the impacts on the state for the crops modeled (U.S. Army Corps of Engineers, 1994).

Arguably, droughts may also incur beneficial impacts in such areas as reducing mosquitoes, lower expenses for snow removal and other related activities. The urge to save water may lead to the uptake of water savings measures and increased commercial activity. Furthermore, drought can help control excess production of agriculture and other sectors, thus contributing to greater price stability and the survival of those agricultural groups that are affected by problems of overproduction. These are not well-researched topics; however, from this perspective one of the proposed strategies to combat drought damage could be to do nothing, considering the benefits, except executing plans to relieve threatened social and economic sectors.

These results help to put the importance of broadening our understanding in relation with all economic impacts of droughts into perspective. This is true not only in relation to direct economic losses to other sectors outside agriculture, but also in relation to other types of costs categories.

Indirect costs

Logar and van den Bergh (2013) define indirect costs from droughts as the consequence of the biophysical impacts on the economy as a whole, that is, through changes in resource based activities on the rest of the economy. The methodological challenge with their identification is with the distinction from direct costs. In this respect, Hallegatte and Przyluski (2010) illustrate a system to help identify indirect losses from extreme weather events (including droughts), they propose the following criteria. First, indirect losses are caused by secondary effects, not by the direct impact of the drought itself. Indirect costs can be caused by hazard destructions or by business interruptions. In addition, costs are indirect if they are spanning on a longer period of time, a larger spatial scale or in a different economic sector than the disaster itself. In their view, this definition avoids consistency problems for slow-onset hazards such as drought. With this definition, the reduction in agriculture yield, and in farmer income for example, are considered as direct costs, consistent with intuition, while the impacts on other economic sector trading with the agricultural sector are indirect costs.

In terms of indicators to measure the extend of indirect costs from drought events, the following indicators have been highlighted in the literature (Logar and van den Bergh, 2013): 1) increased unemployment, 2) changes in prices of food and timber, 3) trade losses and 4) reduced tax revenues, or losses in economic sectors indirectly related to droughts (e.g., food industry). Whilst this list is non-exhaustive, it certainly clarifies that the focus in the calculation of indirect costs from droughts resides in the understanding of the further macroeconomic impacts from the drought event itself. In relation with the estimation
procedure, indirect costs are difficult to ascertain in practice and very often they are estimated with the aid of macroeconomic modeling; e.g. computable general equilibrium (CGE). Below, we select a few studies that have illustrated the indirect costs of drought events by focusing on the presentation of the reported estimates, their comparison with direct costs and outlining methodological approaches for their estimation.

A good representation of the possible economic effects in relation to the estimation of the indirect costs of droughts comes from Australia. Lu and Hedley (2004) report that the 2002–2003 drought in the country led to a significant contraction of the national economy, greater than the relative size of the farm sector alone would suggest. Farm gross domestic product fell by 24.3 percent through the year 2002 to the June quarter 2003, rural exports fell by 26.6 percent, and agricultural income fell by 46.2 percent. Drought related reductions in production also contributed to increased food prices as of mid 2002. The farm sector subtracted around 1 percentage point from GDP growth and around 0.75 of a percentage point from employment growth during the period. These macro-economic effects are large in comparison with the size of the farm sector — typically around 3.5 per cent of GDP but about 20 percent of exports. The drought led to the largest declines in employment on record in the Australian agricultural sector; it is estimated the drought cost the sector around 100,000 jobs, with almost three-quarters of job losses in the grain, sheep and beef cattle farming industries. The size of the decline in employment, in comparison to other droughts, reflects the widespread nature of the drought and the impact of indirect costs (Lu and Hedley, 2004). Furthermore, Adams et al. (2002) and for the same drought in Australia, estimated through the use of the MONASH (GDP and employment forecasts per sector) and TERM (regional CGE) models a total economic impact to the country of -1.6% in GDP (-1% reduction only from agriculture alone) from the previous year and an overall reduction in employment of -0.8% and in the national average wage rate of -0.9%.

For the understanding of the relationship between direct and indirect costs, the revision by Markandya et al. (2010) of the drought affecting the Spanish region of Catalonia in 2007-2008 is a good example. This study estimated indirect costs of around €358.47 million (0.18% of Catalan GDP) related with the decrease on water availability affecting the region’s economy during the dry period. These calculations were done on the basis of an Input-Output model and the analysis of GVA losses to different economic sectors. In this context, indirect costs represent around 60% of the estimated direct costs (see previous section).

Inversely, the economic impact of the 2001 drought to irrigated agriculture in the Yakima Basin in the US was estimated at $130 Million in 2011 by the Yakima Basin Storage Alliance. The agriculture revenue was entered into the IMPLAN models, software dedicated to the estimation of indirect economic loss analysis, and showed that the $130 million direct costs in agriculture revenues contributed to a loss of $196 million in total economic output within the Yakima River Basin region and an estimated total loss of nearly $226 million in total output within Washington State, as well as a temporary loss of 4,900 jobs in 2001 (Yakima Basin Storage Alliance, 2011).

In conclusion, depending on the estimation method and the severity of the drought, indirect costs could be of a much larger magnitude than direct costs. Nevertheless, large indirect costs are often excluded from impact assessments of droughts (Jenkins, 2011).
Non-market or intangible costs of droughts

In addition to the economy, drought also affects the environment and society. The most severe impact is scarcity of water, which affects people, plants and animals. As far as human society is concerned, degraded ecosystems as a result of droughts may result in economic losses, but also to social conflicts and negative impacts on people’s health and safety due to water scarcity (NOAA 2013).

Hallegatte and Przyluski (2010) argue that non-market direct losses include all damages that cannot be repaired or replaced through purchases on a market. For these types of costs, there is no easily observed price that can be used to estimate losses. This is the case, among others, for health impacts, loss of lives, natural asset damages and ecosystem losses, and damages to historical and cultural assets. Sometimes, a price for non-market impacts can be built using indirect methods, but these estimates are rarely consensual (e.g., the statistical value of human life). Unfortunately and based on the controversial nature of the economic methods employed to estimate non-market costs, the literature agrees that these costs are ignored completely in economic assessments of drought impacts (Jenkins, 2011).

Environmental costs of droughts

Environmental costs constitute a major portion of the non-market or intangible costs of droughts. They are results of damages to wildlife (plants and animals) and habitats, as well as of degradation of environmental media such as water, air and soil. The main consequences of severe droughts are felt in aquatic environments, such as rivers, lakes and wetlands. Here, reduced water flows may lead to extensive losses of habitat with consequences for fish and other aquatic species, but also for terrestrial animals which depend on ecosystems in pristine conditions. After all, droughts may cause wildfires, which further deteriorate environmental conditions and may have devastating impacts on society and economy.

By distorting water flows, and with it biological and chemical processes, droughts have a direct impact on the biological communities within an ecosystem. Changes in soil composition may lead to changes in species richness (Wolters et al. 2000) while reduction of connectivity within affected ecosystems hinders animals from moving across their natural habitat. Overall, the resilience of an ecosystem may be severely damaged by droughts (Tilman and Downing 1994), particularly if they occur more often and for a longer period of time.

Droughts may have detrimental impacts on EU environmental policy targets. The European Water Framework Directive (WFD), for instance, stipulates the good ecological status of surface water bodies, which implies the improvement of their physicochemical condition, as well as their flow and continuity (Berbel et al. 2010). Similarly, the Habitats Directive requires the achievement of ‘good conservation status’, which implies the maintenance of the structures and functions of ecosystems. As de la Hera et al. (2011) state, particularly in southern Europe many Natura 2000 sites are surface water bodies and thus prone to negative impacts caused by drought events.

Markandya et al. (2010) state that, while most studies focus on property damages caused by droughts, less focus on higher-order, and intangible (non-market, environmental and social)
impacts. The authors also provide one of the few estimates of non-market welfare losses caused by droughts due to environmental quality decrease. In the case of Barcelona’s 2007-2008 exceptional drought event, they estimate that non-market welfare losses related to the decrease of the ecological status of the river basin due to the lowering of water flows to €167.76 million or 0.08% of Catalan GDP. These costs have been estimated on the basis of value transferability of a stated preference valuation of environmental costs in a nearby river basin.

By estimating people’s willingness to pay (WTP) for continued river flows, the socio-economic benefits of appropriate policy measures can be assessed. In this context, Berbel et al. (2010) show that the population of the Guadalquivir River Basin in Spain derives significant benefits not only from the direct use of water, but that also holds non-use values related to the ecological status. They estimate that average WTP for the allocation of water to the river basin for the maintenance of environmental services amounts to €7.95 and €10.88 per household per year, for an improvement to a good and a very good ecological state, respectively. Average WTP values were calculated through the application of choice experiments using the Conditional Logit Model. Similarly, Alcon et al. (2012) applied the contingent valuation method to estimate the non-market environmental benefits of using reclaimed water to maintain river flow levels in the Segura River Basin in south-eastern Spain. The results show that the average household was willing to increase the amount paid for waste water treatment by €5.31 (from currently €6) per month or €63.72 annually.

While there is only limited evidence on the economic costs related to ecosystem damages and decreased environmental quality as a result of droughts, the available literature shows that costs can potentially occur on two levels. First, environmental costs might occur as a direct result of water scarcity, e.g. the reduced water flow in rivers and streams. Second, deteriorated water quality as a result of higher temperatures and reduced water availability might also result in environmental costs, e.g. the loss of biodiversity and ecosystem services. As Markandya et al. (2010) have shown, studies estimating the WTP for continued river flows (e.g. Berbel 2010, Alcon 2012) can form the basis for a value transfer exercise valuing the on non-market or intangible costs of drought events based on the forgone benefits of continued river flows.

**Social costs of droughts**

As introduced previously, drought impacts are usually grouped into three principal areas: economic, environmental and social. Arguably, direct and indirect costs take account of financial economic losses, whilst social and environmental losses are of a non-market economic cost nature. From an economic perspective, social impacts can be of a significant magnitude in relation with the other cost categories and can illustrate severe economic consequences. In this context, the focus of the drought impact analysis would be to account for any welfare changes experienced by human beings. Quantification and monetization of social costs is seen in the welfare economics literature as a basic requirement for sustainability and a precondition for achieving economic efficiency in public investment (Markantonis et al., 2011, 2012). In order to integrate all environmental damages in an integrated assessment and management of drought risks, these benefits need to be valued in monetary terms (Pearce and Turner, 1990).
Following with the cost categories developed under the CONHAZ project, Health and social costs of droughts refer to:

1. Health costs
   - an increased risk of diseases,
   - malnutrition and famine due to food shortages.

2. Social costs of droughts involve
   - loss of human lives,
   - migration (usually from rural to urban areas),
   - social conflicts,
   - changes in income distribution, and
   - social welfare losses due to restrictions of water supply in households (e.g., prohibition of water use for swimming pools, gardens, or car washing).

This section focuses on the understating of the non-market social cost categories that are more relevant for the case study of the Tagus-Segura, which mainly come down to the understanding of the social impacts from water restrictions and a reference to changes in income distribution from droughts.

Water restrictions usually constrain particular uses of water, but not necessarily require households to reduce the amount of water they use per se. Therefore, water restrictions are aimed at limiting householders’ freedoms regarding water use and do not directly address the fundamental issue of ‘total use of water’. Imposed water restriction can be temporal (e.g. only a few hours a day) or specifically restrict certain uses/activities in times of water scarcity (e.g. gardening, swimming pools). Understandably, there is substantial conjecture about the positive and negative consequences on the use and application of restrictions in the eyes of consumers. In this respect, several studies have researched with the aid mainly of stated preference methods the WTP to avoid water supply restrictions or WTA for a change in supply reliability. Ideally, water allocation under scarcity should follow a simple rule: Those users who gain the highest marginal social value from use should have priority (Winpenny, 1994). This “economic” rule is often not given priority in water management, with social and political objectives generally dominating decision making.

Many examples exist (mainly from the US and Australia) that have estimated WTP values to avoid water supply restrictions. Cooper et al. (2011), in a contingent valuation study that investigated consumers’ willingness to pay to avoid urban water restrictions in Australia, found out minimum and maximum WTP values ranging from $98-$291 per household year depending in differing income characteristics and other variables (e.g. water rich city or owning a lawn). The identification of higher value water uses is fundamental in understanding the variation in value estimates. For examples, respondents with a lawn had on average a WTP of $152 compared to those without whose WTP were around $98. Participants with a higher income indicated a WTP value of $181, with an upper bound WTP value estimate of around $291. Alternatively in Camberra (Australia), Gordon et al. (2001) carried out of a choice experiments study with the aim of estimating residents’ WTP to avoid public water supply restrictions. Results suggest that residents were willing to pay, on average, a very small amount ($10) to prevent a 10 percent reduction in water use (in 1997 Australian dollars). On the issue of the impact on the urban environment, residents indicated that they were willing to pay $18 per annum to improve Camberra’s general urban
appearance, from ‘brown’ to ‘some brown’, although WTP for further improvements – from ‘some brown’ to ‘green’ – was found to be insignificant.

In the US, Griffin and Mjelde (2000) examined customers’ preferences in seven Texan cities, using contingent valuation methods. Respondents were found to be willing to pay, on average, between $25.34 and $34.39 (in 1997 US Dollars) to avoid an occurrence of water restrictions. They also found, however, that respondents were willing to pay, on average, $9.76/month (or 25.6 per cent of their bill) to improve future supply security levels. In addition, Koss and Khawaja (2001) carried out a contingent valuation study of householders’ valuation of system security in 10 Californian water districts. Respondents indicated that they were willing to pay, on average, between $11.67 and $16.92/month to avoid restrictions (in 1993 US dollars), depending on the frequency and severity of the restrictions.

In contrast in the UK, the water utility Yorkshire Water, commissioned a choice modeling study in 2002 (Scarpa et al.; 2004). Four service levels were included in the experiments: Existing service levels (where the expected frequency of having no water was once every 500 years); a decrement in service levels (once every 250 years) and two levels whereby service was improved (once every 750 and 1000 years, respectively). They found that residential customers were willing to pay only about £0.20 on average for a one level improvement in reliability (e.g. to reduce the expected frequency of having no water from once every 500 years to once every 750 years). Business customers indicated that they were willing to pay about £1.74, on average, for a one-level improvement.

In terms of WTA compensation for urban water supply restrictions, Howe and Smith (1994) conducted a contingent valuation study of residents in three towns in the United States: Boulder, Aurora and Longmont where it was proposed due to water scarcity that residential outdoor water use would be restricted to three hours every third day for the months of July, August and September. The study found that between 41 and 58 percent of respondents indicated that a decrease in supply reliability, even if accompanied by a change in the water bill, was undesirable. Furthermore, of those willing to consider a change in supply reliability, respondents indicated that they would accept between $4.53 and $13.99/month (in 1994 US dollars), on average, for a decrease in supply security.

In the city of Seville in Spain, Garcia-Valiñas (2006) analyzed the impact of water use restrictions and water quality reductions on consumer welfare during the drought period of the early 1990s. The water demand functions were estimated for residential and industrial/commercial customers, respectively, using quarterly water bill data as well as other economic information. The welfare variations were then calculated based on the water demand functions. The average welfare losses, in 2001 Euros, were €138.3 ($124.5 in 2001 US Dollar) per user and quarter for households and €62.6 ($56.3 in 2001 US Dollar) for industrial/commercial firms.

These studies above show wide variations in WTP/WTA. Henscher et al., (2006) conclude that the reasons are unclear and wonder that in part they are likely to be linked to the methods used and in part to the specific trade-offs being targeted in particular countries associated to specific cultural uses of water, which differ across international studies. In this respect, it is evident from the literature that the public has a strong preference and place a value in the avoidance of water restrictions as a tool to reduce the impact of droughts. In terms of the
assessment of drought impacts, these welfare losses can account for a large proportion of the total costs and somehow need to be investigated.

In a different context, we have previously introduced that declines in water availability and irrigated area have an effect throughout the agricultural economic system. Some of the non-market consequences relate to an assessment of changes in income distribution. This category is very much linked with the analysis of indirect effects from droughts. Changes in wealth distribution are often ignored in drought assessments and their effect is not well understood. In the agricultural sector for example, the effect of droughts to local farmers may decrease their ability to repay loans from local moneylenders, who very often also serve as sellers of agriculture inputs. Toulmin (1987) highlights in a report for the FAO that the ability of farm households not to pass on to others the effect of a drought period will differ according to the following: 1) their ownership of assets, 2) their access to incomes from other sources and 3) the extent to which these assets and incomes are less affected by drought than are from harvests. Fundamentally, the assessment of income distribution will entail an analysis of the efficiency of production at the farm gate and other businesses which are impacted by droughts. There are instruments in place, such as farm insurance, that can help affected parties to sail the storm. Nevertheless, as seen previously with the incidence of direct costs, the impact of droughts are normally felt at the local level, changes in income distribution will show geographical changes of income between the affected area as opposed to unaffected ones. This cost category is important for the development of strategic actions to reduce drought impacts, as these strategies would have to account for lost competitiveness of efficient producers that lose their market power as a consequence.
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Research Task 4.2 Output 06

The institutional potential for water markets in the Tajo and Segura basins

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1. Introduction

Water scarcity is a fact in the Segura River Basin (SRB) and increasing in the Tajo River Basin (TRB). These basins need to address scarcity. Their connection via the Tajo-Segura inter-basin Transfer project (TST, capacity 600 hm3 per year) means that their strategies and actions can affect each other. Deliverable 4.2 has background information on the physical and institutional characteristics of the SRB, TRB and the TST.

Most water in these basins is used by farmers for irrigation, but demand from municipal, industrial and environmental sectors is increasing. We take those demands as given and concentrate on managing the impact of scarcity on farmers. From the supply side, scarcity can be reduced by providing more desalinated or recycled water. From the demand side, scarcity can be reduced through drought insurance (farmers take money instead of water) or irrigation efficiency (assuming that farmers use LESS water for the same volume of crops). All of these actions can be complemented by markets that make it easier to reallocate limited water among existing users.

For background information on water conflict in Spain, see Alarcón, Berbel, Candela and Domingo (2008). For detailed information on markets for water in Spain, see Garrido, Rey and Calatrava (2012) and Rey, Garrido and Calatrava (2012) on how they have functioned, their limits, and suggestions on how to improve them.

We know from these works that auctions for water (a particular kind of market) have not been used in Spain. After describing the institutional conditions in the SRB, TRB and TST and how they function (or not), we review the preconditions and benefits of markets and/or auctions in the the SRB-TRB-TST system (TST).
2. Some details on how water is currently managed

The general situation of water scarcity is increasing in TRB and SRB due to increasing demands for water and decreasing supply (higher irrigation efficiency reduces runoff; groundwater levels are dropping; and environmental needs are rising). Existing management faces the difficult task of trying to manage this scarcity while sometimes obeying laws and responding to traditional views on how water should – or should not -- be allocated. Water users have engaged in many schemes to improve water allocations (e.g., informal, internal trading), whether or not these schemes are legal or conducted in public. Some users (especially in TRB) have found ways to be simultaneously in need of water (asking for larger allocations) and overendowed (selling water within or across basins). Overlapping (and conflicting) promises to various groups are harder to reconcile; new changes (lack of money; desire for environmental flows) make conflicts more visible by removing “ad-hoc allocations” of money or water from the basket of management tools. The situation, in short, is getting too complicated to make any sense to outsiders. Social, traditional, environmental and other conflicting and overlapping “needs” are too complex to comprehend or reconcile under current management norms. It would be useful to transition from the current situation to one in which priorities are clear (i.e., social and environmental flows before economic flows) and allocations rational (i.e., allocation of economic water via prices and markets).

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1 This section – more than others – relies on information gathered in May 2013 conversations with Julio Berbel, Javier Calatrava Leyva, Nuria Hernandez-Mora and Adolfo Mérida Abril. Also see Annex 1 for their subsequent response to an earlier draft of this paper.
3. Preconditions for water markets

Wholesale water prices in TRB and SRB generally cover costs. TST prices paid by SRB farmers cover more costs than “usual” for infrastructure projects in Spain (100% of CapEx and 50-60% of OpEx). It may not be feasible to raise these prices to reduce demand, as some farmers are inelastic at current prices - meaning that higher prices just “take farmers’ money.” Farmers with higher elasticity (growing cereals instead of olives or greenhouse crops) may respond to higher prices, but then what do you do with the water? Allocations need to be formalized as rights before management can affect water use allocations or efficiency.

Let me note here that there is a consensus that water users associations (WUAs) are well run, there’s always a need for more meters, and that groundwater management is improving.

The 1985 groundwater law needs to be fully implemented; it’s widely acknowledged that groundwater in SRB is under stress, “because it’s profitable to take more.” Given this reality, it would be unwise to implement a market for water due to the predictable negative impact of such a market on common pool groundwater resources. As we have seen everywhere in the world with weak control of groundwater use, a market for surface water means that surface exports are replaced by greater pumping of groundwater.

These latter two characteristics need to be strengthened before allocations will work well. Drought management plans are useful in documenting and coordinating those efforts.

Even assuming rights, existing laws, administrative codes and social mores make it difficult for farmers to trade water because, respectively, intersectorial or interbasin trade

\[\text{These percentages are controversial, as there are many ways to calculate cost share. It’s likely that the real price they face is lower.}\]
is restricted or blocked, “excess” waters may revert to WUAs, and farmers worry that they will lose rights to the public’s water if they do not use it. Laws, codes and mores would need to change. Laws can change if allocations are recalibrated to consider environmental flows and place the same “stress” on each basin. They would also need to be simplified, to remove sectorial (i.e., agricultural or urban) restrictions and account for “carriage losses.” Once they are changed, administrative codes would need to be changed so that water rights (or licensed flows) could be severed from land. That revision would change land values by dividing current values into “dry land” and “water rights.” It would also require that WUAs separate and secure liens against those rights, in proportion to their values and/or connection to WUA costs. Social mores may be the slowest to change, but they will be more flexible if the social dimensions of water (e.g., environment, human rights, poverty) are addressed at the same time as its economic dimensions, i.e., it’s easier for citizens to accept water trades as something similar to land trades when they know that water’s other social functions are respected.

4. The social dimension

Most water users have a traditional right that they have not paid for or for which they have received subsidies for use. These facts make it hard for people to accept that they should be allowed to make a profit for using water that belongs to all (per the Spanish Constitution). Chile has dealt with this problem by granting rights without condition; Australia has tried to do the same but found that environmental flows were excessively low and/or that cities were not able to buy water (building desalination plants instead). Spain might follow modified versions of Chile or Australia’s system – or design its own rights’ system – but it seems clear that existing owners must lose some water or some money in exchange for rights, as a repayment to the people for their water.
5. Existing markets and trades

Farmers now trade water among themselves in the SRB and TRB (legal or not). In addition to basic TST transfers, additional TST trades occur when conditions allow (“drought emergency”), at negotiated, private prices. Some farmers are looking into an options market for TST water.3

The existing basic TST transfer (averaging 290 HCM – 60/40 to agriculture/urban -- but not above 540HCM and not to the TST’s capacity of 1,000HCM) is based on a declared surplus that may not respect reality.4 That allocation -- and its use of administrative prices rather than prices determined by supply and demand -- reduces support for markets by creating the perception that trading means moving water without reference to scarcity at prices that may not exceed willingness to accept in the TRB.5 This fact, combined with the negative environmental impact of a trans-basin transfer, means that the TST gives a bad name to “markets” that may function (and apparently do function) very well as the local, basin level. It may be hard to reform the TST using administrative tools (i.e., claims) that cannot be reconciled. Markets – and the TST auction described below, in particular – may offer an alternative way to change TST flows in a way that all sides may accept as fair.

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3 This idea may attract opposition to the extent that it signals water markets as a normal economic activity, instead of being an “emergency to save the community.”

4 Environmental flows were not considered when the TST allocation was declared. Voluntary reallocations from TRB in drought may have detrimental environmental impacts because they are allowed within formulas that place only superficial weight on environmental flows (e.g., flows need to be in some locations, not along the entire river bed).

5 TST flows during drought have a negative environmental impact; they also take water from cities like Madrid that would love to get them – or buy them. Note that there are provisions to prevent TST transfers under certain conditions, but these conditions do not necessarily match either e-flow needs or supply/demand conditions in the TRB. It’s ironic but instructive to consider that a drought declaration that prevents TST transfers at administrative prices may result in SRB-TRB trades at mutually-agreed volumes and prices. Those trades may, unfortunately, have negative impacts in the TRB, if sellers manipulate local accounting or allocations of water rights to “free” water for interbasin sale and export that should have been left for in-stream flows.
6. Why use markets or auctions?

Markets are useful for helping buyers and sellers make mutually beneficial trades. They require the conditions discussed above if they are not going to create problems. Auctions take more time to set up than markets, but they can be good for coordinating numerous trades that need to be cleared through physical infrastructure (canals or pumps) and establishing a single price that reflects and balances all existing beliefs on supply and demand. Markets increase efficiency, and thereby increase wealth. That wealth (or the potential of accessing it) can be used as a reward for undertaking the reforms necessary for a market, e.g., paying for meters, reducing illegal groundwater use, accepting a reduction for environmental flows, etc. Auctions are better than spot markets (where buyers and sellers occasionally interact) because they clear as many trades as possible at a common price. Such transparency also makes it easier to coordinate water delivery at the close of each auction, an action that’s more difficult with bilateral trades that occur over weeks or months. Note, of course, that occasional auctions (once per year, month, etc.) do not prevent bilateral trading in markets.

Markets or auctions (interchangeable here) also improve operations and stability at WUAs, because they do the “hard work” of balancing supply and demand and setting a price for water. WUAs definitely need to establish a means of covering fixed and operating costs in the existence of a market (by, e.g., taxing land or charging severance fees and collecting a carriage charge, respectively).

From a larger view, markets improve the social allocation of water by clarifying its price and value. SRB and TRB water users now use water based on some combination of administrative, political, economic and shadow costs. Some prices are set by bureaucrats, some water is off limits or mandatory in use, some water is traded transparently, and
other water is taken at a cost to others. A market, for example, for ground, desalinated, TST and local surface water in the SRB would have a single price that reflected transaction and operational costs more than artificial constraints, subsidies and asymmetric information. Such a market would not just improve water allocation efficiency; it would also reveal the economic value of various sources of supply. Markets for water in TRB would reveal the values of water in use (or their lower limits) as well as move water to higher values. Market signals may make it easier to reduce friction over allocations among farmers, cities and the environment.

7. Auctions in the SRB and TRB

It’s less complicated to start with separate auctions (or markets) in the SRB or TRB, as intrabasin auctions do not affect resource allocations in the other basin. Auctions of markets in either basin would take place after establishing preconditions, i.e.,

- Defining how much surface and ground water is available
- Allocating rights to that water
- Reserving water for storage or environmental flows
- Dividing water into urban and irrigation pools
- Allowing markets or auctions for irrigation water

It’s politically controversial to put urban water up for sale (even if farmers cannot afford it) as well as technically challenging. Ignoring issues of water quality, most urban water is managed by utilities that want to control demand and prices with traditional methods (effective or not). In short, it makes more sense to start with auctions for

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Desalinated water – now overpriced and underappreciated – would be easier to sell, at the right price, instead of being “gifted” to some group at an unacceptable cost.
irrigation water because farmers use more water, are more willing to buy or sell water at market prices, and share water distribution networks.

Auctions of known quantities of water in either SRB or TRB could take place in two ways. The first would be a two-sided auction (essentially a market) in which sellers with known rights to a quantity of water (as opposed to a permanent rights to the flow of water from a source) offer that water at “ask prices” that buyers could accept. Buyers can also offer “bid prices” that sellers can accept. This system results in lots of asks and bids and prices that vary, but it tends to move the water from sellers to buyers fairly quickly. The second method would pool the water and allow bids from farmers who want the water (a pooled auction). High bids would get the water, but all bids would pay the same price (equal to the highest rejected price). Revenue from the auction would go to those who had rights or the basin organization with the responsibility of distributing water and recovering costs, as appropriate; see Zetland (2013).

Auctions or markets within basins allow efficient water reallocation while recognizing rights and placing a price on the water that’s closer to its value. Auctions can be used for greater or smaller quantities of water, as often as necessary, and with as many participants as local stakeholders prefer. The prices from auctions provide a useful signal of willingness to pay/accept, which helps with other water management issues. Note, of course, that rights in auctions need to be clearly specified. As with markets, it is not a good idea to have an auction for some rights if sellers are going to replace sold water with water taken (illegally) from uncontrolled, common pool sources (i.e., groundwater from aquifers).
8. TST auctions

Auctions would be better than markets for connecting the SRB and TRB, as a means of identifying how much water to transfer, who will get less water (in exchange for money) and who will get the transferred water. It’s not clear whether an TST auction should be run simultaneously with SRB and TRB auctions or markets. A simultaneous auction would clarify how many units would be transferred from TRB to SRB as well as who would get water in each basin, but it would raise average prices in TRB and lower them in SRB. Such an outcome may be politically unpopular in TRB. Sequential auctions, on the other hand, would result in three sets of prices, one for each within basin auction and one for the TST auction. Sequential auctions would allow sellers with a “sudden” surplus of water to sell to buyers with a “sudden” demand. Prices in a sequential framework are likely to be volatile, as these supplies and demands may not line up very well. Higher (pre conveyance) prices in the TST would also create “windfall profits” for TRB sellers, which may be unpopular. As another choice, it is possible to set aside a fixed quantity of TRB water that would be sold at auction to TRB buyers (in a single-purpose auction or auction that included all SRB waters), with revenue going to the TRB authority. TRB could then allocate remaining water in an auction or administratively. These pros and cons must be discussed and settled by stakeholders, as the final form and timing of auctions need to suit local preferences.

9. Stakeholder perspectives on markets

The Annex includes comments on an earlier draft of this paper. These comments sometimes conflict with each other or with my views as author; some of them have been

7 SRB prices would be lower for the subset of users with access to TST flows; other users may also face lower prices due to a reduction in demand from users buying TST water.

8 SRB prices must include the cost of conveying water, e.g., EUR 0.20 per m3 includes EUR 0.13 per m3 conveyance cost, per p 31 of Del 4.1.
integrated into the corrected text. They are worth reading as a means of understanding the nuances of water allocations and marketing in SRB, TRB and TST.

10. Summary

Markets and auctions are useful in matching supply and demand in a quick and transparent way. They can be used locally (among irrigators in one water users association), within a basin, or across basins. Markets in the SRB and/or TRB, accompanied by an TST auction could efficiently allocate water from many sources and seller, provide useful price signals, and leave enough surplus left over to increase environmental flows and pay for system costs and debt. Markets or auctions, however, should not be used in the SRB, TRB or TST until groundwater levels are managed, water rights are clearly defined and unified, and the social impacts of water trading are addressed. As a first step, I suggest ignoring TST trades for the moment and concentrating on TRB and SRB intrabasin management of water and money. Once those basins are working, it will be easier to reform TST trades and pricing.

Wholesale water prices in TRB and SRB generally cover costs. TST prices paid by SRB farmers cover more costs than “usual” for infrastructure projects in Spain (100% of CapEx and 50-60% of OpEx). It may not be feasible to raise these prices to reduce demand, as some farmers are inelastic at current prices - meaning that higher prices just “take farmers’ money.” Farmers with higher elasticity (growing cereals instead of olives or greenhouse crops) may respond to higher prices, but then what do you do with the water? Allocations need to be formalized as rights before management can affect water use allocations or efficiency.
Let me note here that there is a consensus that water users associations (WUAs) are well run, there’s always a need for more meters, and that groundwater management is improving.

The 1985 groundwater law needs to be fully implemented; it’s widely acknowledged that groundwater in SRB is under stress, “because it’s profitable to take more.” Given this reality, it would be unwise to implement a market for water due to the predictable negative impact of such a market on common pool groundwater resources. As we have seen everywhere in the world with weak control of groundwater use, a market for surface water means that surface exports are replaced by greater pumping of groundwater.

These latter two characteristics need to be strengthened before allocations will work well. Drought management plans are useful in documenting and coordinating those efforts.

Even assuming rights, existing laws, administrative codes and social mores make it difficult for farmers to trade water because, respectively, intersectorial or interbasin trade is restricted or blocked, “excess” waters may revert to WUAs, and farmers worry that they will lose rights to the public’s water if they do not use it. Laws, codes and mores would need to change. Laws can change if allocations are recalibrated to consider environmental flows and place the same “stress” on each basin. They would also need to be simplified, to remove sectorial (i.e., agricultural or urban) restrictions and account for “carriage losses.” Once they are changed, administrative codes would need to be changed so that water rights (or licensed flows) could be severed from land. That revision would change land values by dividing current values into “dry land” and “water rights.” It would also require that WUAs separate and secure liens against those rights, in proportion to their values and/or connection to WUA costs. Social mores may be the slowest to change, but they will be more flexible if the social dimensions of water (e.g., environment, human rights, poverty) are addressed at the same time as its economic dimensions, i.e., it’s easier
for citizens to accept water trades as something similar to land trades when they know that water’s other social functions are respected.
References

Annex 1: Stakeholder comments

I contacted several key stakeholders and experts to get their comments on an earlier draft of this report. They are included here, with some small edits for clarity. Some comments have been integrated into the current version, but some were not -- either because I did not agree with them, or because they conflicted with others’ views. These comments therefore clarify important and controversial facts and perspectives regarding the SRB, TRB and TST.

Adolfo Mérida Abril (28 May)

First of all, as a general view, the document is more or less focused on, let’s say, the mismanagement of the transfer as the reason why it should be revised. I agree in the fact that some revisions should be carried out, but I think it is also fair to say that TST is critical to meet urban demands, especially during the last drought, and it also enables a very profitable agricultural activity. Irrigation associated to TST is probably the one with the highest technology of the world. I prefer to see the TST as a good thing that could be even better with some revisions.

Going down to the detail, some points I’d like to highlight:

- Regarding “The existing basic TST transfer” (page 3), the 600 hm3 mentioned are associated to the Tajo abstraction; Segura is supposed to receive as much as 540 (140 for urban supply and 400 for irrigation). On average, we have received 288 hm3, of which 113 were for urban supply and 174 were for irrigation.

- I totally disagree in the description of the price of transfers; it is administrative, but I think it is not accurate to say that it is ‘subjective’ and ‘negotiated’ when there is an specific law and several regulations that describe how to set the price and how to update it. You may agree or not with the procedure to determine the price, and its updates, but the procedure follows certain rules that can’t be ignored, so it is not ‘freely’ negotiated or ‘totally’ subjective.

- I don’t agree either with the statement that says that the transfer happens ‘regardless of scarcity’. One of the principles of the Transfer is the respect to the Tajo demands, so certain limits are applied to the maximum rate of transfer. There are different levels of alarm, from normality to no transfer permitted, depending on the Tajo situation. In fact, when there have been problems in the Tajo basin, the transferred volume was lower than the maximum volume allowed according to the alarm level. However, I agree with the fact that the surplus doesn’t respect reality and it has to be updated. As a conclusion, I would say that alarm levels have to be revised to better protect Tajo demands, and in fact a revision is proposed in the current Tajo Management Plan draft.

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• Just a thought. If one of the reason why the TST must be revised is that there are scarcity problems in the Tajo basin, I don’t see how a market will improve that situation.
• Finally, I would like to add just a tiny comment to the summary. I guess that the considered TST auction is not constrained to current TST users in the Segura. Currently, the users that receive water from the TST are perfectly identified and geographically located. They are somehow isolated from the rest of the Segura basin users, though the TST users also receive Segura river basin resources. From my point of view (just my opinion), TST auctions could work regardless a modification of the intrabasin management if TST users are treated independently from the rest of users (like it happens with trades during droughts).

Nuria Hernández-Mora10 (28 May)

Some general considerations:

• I think one consideration that is only marginally mentioned in the paper are the environmental requirements which are more often than not overlooked when setting up market mechanisms for water allocations. In order to prevent running into problems the environmental requirements should be front and center of any review of allocation mechanisms. This is particularly relevant in the case of the Tajo and Segura basins where the transfer has precluded any scientific consideration of environmental flows for the Tajo.
• I think it is important to acknowledge that the rules of the TTS are problematic and conflicting. The fact that most transfer decisions have ended up in the Spanish supreme court because of processes initiated by either the Municipal associations of the upper Tajo Dams (Asociación de Municipios Ribereños de Entrepeñas y Buendía) or/and the Autonomous Community of Castilla-La Mancha (the one that is most directly impacted by the transfers) is indicative of these problems. Most demands have not prospered, but that does not mean the conflict does not exist.
• Also, we need to be aware of the tremendous political pressure that exists as a result of the existence of the TTS. Any institutional change in water allocation rules needs to take this existing pressures into account. The proposed Tajo basin management plan (BMP) and the management of the 2005-2008 droughts are good examples of this political limitation:
  o The Tajo BMP only proposes minimum flows for strategic points in the basin. They explicitly say they do not propose environmental flow regimes (as is done in other basins in compliance with the law) because doing so would make the transfer no longer viable.
  o When markets (contratos de cesión) were allowed between the Tajo and Segura basins during the drought two things happened:
    1. Water allocation to the selling irrigator communities in the Tajo (Estremera) were doubled in order to allow for the sale of more water and thus override transfer limitations in times of drought.
    2. Purchase agreements between Tajo irrigator communities (Canal de las Aves) and urban users in Alicante and Murcia (taibilla) where agreed upon

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to a large extent in order to avoid water levels in the donor basin dams (Entrepeñas and Buendía) to fall below what Adolfo calls the "emergency levels" which would have prevented further transfers (much cheaper for users than the purchased water through markets).

This is important because rules, no matter how well designed, are often "tweaked" when strong economic and political interests are at stake. Therefore transfers do occur regardless of scarcity. During the drought stretches of the Tajo dried up (for example Talavera in the summer of 2006) and municipalities in Castilla La Mancha suffered restrictions, but the transfer continued.

**Javier Calatrava**\(^\text{11}(29\ May)\)

It’s been 15 years discussing many issues around the TST in a lot of forums and meetings and here we still. It is very unlikely that we could have a really insightful discussion by e-mail.

I will thus focus on the TST water markets issue and try to be as brief and schematic as I can in exposing my view:

- Both Nuria and Adolfo are right in the evidences they expose but I do not fully agree with their conclusions or yours.
- There are a lot of water management issues to be addressed before having efficiently working water markets. I guess we all agree with that.
- These issues should be addressed both in the Tajo and Segura basin (and a few other basins in Spain) and with equivalent rules (e.g. monitoring of uses, criteria for defining crop's water requirements, pricing schemes, cost recovery, urban wastewater treatment standards, environmental requirements, etc.) for all players. Do we all agree on that?
- These issues should be addressed regardless of the existence of water trading. I guess we all agree with that.
- If these issues not been properly solved implies that inter-basin water trading cannot take place I do not understand why intra-basin within each basin should be allowed. If pre-conditions for an efficient TST water trading to take place do not hold, they do not hold either for any other water market. The question is thus "water trading yes or no?" rather than "TST water trading yes or no?". I guess we do not all agree on that.
- I think it’s possible to "fix" conditions in one basin -- allowing intrabasin trade -- without the other basin, so that’s why intra may proceed inter. TST can only work when conditions allow in both basins...
- By the way, my answer is "yes to the option of water trading" both within basins and between them, not only in drought periods, with a proper objective assessment of impacts (much better than done in the past) and no political interference.

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• Last, if we wait for these problems to be solved before having water trading we will probably never have any water trading and lose the flexibility they provide to water allocation. Being practical, and in absence of a reallocation of water concessions by the Administration itself, I opt for allowing water markets even if they are not fully efficient and progressively work to advance in easing trading and improving their monitoring. I guess you will not all agree with me.

I tend to agree with you, but there are real and imagine problems with markets when, e.g., groundwater or e-flows are not addressed. Third party impacts (reduction in production and therefore, e.g., seed sales) are not so relevant, but markets can make conditions worse if fundamentals are not fixed -- leading to either harm or a backlash against markets that shuts them down for decades (see, e.g., Owens Valley vs. Los Angeles -- a trade that is STILL causing problems 100 years later...)

Going to more practical questions, Nuria indirectly points at something I have been defending for quite a lot time. Water allotments for irrigated areas in the Tajo are quite generous (clearly above quota on the 2005-2008 trading with the Segura as Nuria says and as it is clearly written in the Tajo basin 1998 Hydrologic Plan). I agree with Nuria in the need to secure more river flow "along" the Tajo. But we probably disagree in whether only the TST users or also other water users in the Tajo should contribute to that. If there is a conflict to be solved, both parties should contribute. Strong oppositions in both directions do not solve conflicts.

That's why markets are better, because they turn "forced contribution" into "voluntary trade" -- we will find out who contributes when they sell into the market. E-flows can be bought in the market, be accidental due to downstream purchases, or be set-aside before markets allocate remaining water. It's consistent to set aside e-flows within a basin, then look into reallocation within/outside of basin...

Water trading between the Tajo and the Segura solved important problems but put additional pressures on the Tajo. My doubt is whether a part of those pressures (large or small) can be reduced by a better water management in the Tajo. If Talavera had water restrictions, the blame is not to be put on the trading with the Segura but rather on the Tajo Basin Authority giving priority to agricultural users (against the law? was there any sue in the courts?), regardless of whether they were later trading with that water or not.

This Insha'allah question remains to be seen, and it's outside the scope of my report (or capabilities).

**Julio Berbel**¹² (29 May)

**General comments: laws and norms.**

I don’t agree that laws are outdated. Water Law has been revised and adjusted to Water Framework Directive (2000). In general terms is a modern law adapted to difficult Spanish conditions. It is the only law in Europe that allows water trade and I believe that it goes beyond WFD in many issues.

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Drought management plans are necessary instrument for managing the usual "drought period" that unfortunately happens with a frequency of 2 out of 10 years, but they are restricted to those period, rest of time normal condition holds (scarcity).

Scarcity is linked to Mediterranean basins and is natural condition. Drought is a recurrent frequent situation.

Drought declaration in Tajo basin implies the limitation or restricted transfer (2 out of 10 years)

Regarding affirmation that “excess” waters revert to WUA, this is not a serious problem (in my opinion). Some WUA are introducing ‘internal trade schemes’ to trade water inside the WUA, and those experiences are working well.

The present Spanish regulation may be sufficient for starting market creation (within the existing legal framework but with improved governance and transparent operation). But the question is: why there are so few exchanges then?

The TST

National policy (Parliament) has interconnected two basins Tajo and Segura through TST and therefore they should be treated as a single entity, as you say “Laws can change if allocations are recalibrated to consider environmental flows and place the same “stress” on each basin.”. By the way ...what is the limit of a basin / sub-basin? (it an arbitrary human-made definition)

The TST allows a ‘cheap’ transfer as buyers pay to the government the transport cost (including some environmental compensations). Only drought declarations in Tajo basin forces inter-basin market into operation as the administrative allocation TST is restricted.

The markets

In general, markets operate when there is scarcity and there is a value differential higher that transaction cost. I believe that this is the case in Segura basin but it is not the case in Tajo as water allocations are oversized. Market is already operating in Segura at certain level but I believe that is not doing and will not do in the near future in Tajo unless the basin is ‘closed’ (no more abstraction permits) and water allocations are more justified.

If the Parliament (where national sovereign is located) decide to close the TST, it is OK, but I don’t agree with the use of instruments such as definition of environmental flow in Tajo river as a tool against the TST. (Note that European Commission has not decided yet a common procedure -e.g. see Blueprint for water-).

The future

I believe that maybe we need a single authority in < Tajo/basin joint-basin>. Then the question will be to decide a rational definition of environmental flow and reserves and let the economic uses in the “joint Tajo/Segura basin” to be self regulated by markets (when all pre-conditions that Javier suggest are reached).

Auctions can be an instrument among others (Exchanges such as Australia, water banks, etc).
Research Task 4.2 Output 10

Mirror CS in the Reno river: potential for water use right market development

17th April, 2013
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1. The Po River Basin

With 71,000 km² (~24% of the state territory), the Po river basin is the largest (single river) basin in Italy and the economically most important area. The basin area is home to 17 million inhabitants (~28% of the state population). More than one third of country’s industries producing 40% of the national GDP are located in the basin area. The agricultural output accounts for 35% of the national production. The agricultural sector generates an added value of about 7.7 billion €/year (~1.2% of the total added value produced in the basin). The one thousand or so hydroelectric plants installed on the Po River and its tributaries generate on average 20 billion kWh/year (~48% of the installed hydropower in Italy). Additional 400 thermoelectric plants generate around 76 TWh every year. The natural and artificial lakes in the basin regulate a volume of 1,858 million m³ per year (AdBPo, 2006).

The river basin spreads over eight (out of twenty) Italian regions including Valle d’Aosta, Piedmont, Lombardy (all three entirely included in the basin area), Emilia Romagna (with about a half of the area included in the basin), Autonomous province of Trento, Veneto, Liguria and Toscana (marginally included in the basin area).
Po river basin annual average precipitation is 1,108 mm with maximum values in the Alps, over 2,000 mm per year and minimum in the eastern Paduan plain, 700 mm per year (AdBPo, 2006). This amount of precipitation produces an annual water flow of 78 billions m³, which correspond to a water flow of 2,464 m³/s. Two third of this flow runs on the surface, that is approximately 47 billions m³ per year, 1,470 m³/s. The remaining 31 billions m³ are consumed by evapotranspiration and deep percolation. Two mountain chains, Alps and Apennines, feed all rivers in the basin. River cycle characteristic depends on the source of water. Alpine rivers have water flow peak in summer due to ice melting, while Apennines’ rivers have lowest peak in summer due to their dependency from precipitations, and highest peaks in spring and autumn.

Po river basin is water rich thus its surface water component is remarkable. The principal reticulum includes 141 major water affluents (>20km of length), while the secondary surface river network is nine times more extended than the primary river network, which lengthens in the basin for over 6,750 km (AdBPo, 2006). Artificial
networks, including irrigation channels and drainages, are also highly developed throughout the basin. This complex and extended water network is the result of thousands of years of human alterations of the natural environment. Flow of water from mountain basins and natural lakes to the Po river running along the Paduan Plain is intensively interfered by artificial abstractions, rice field submersions, dripping irrigation, deviations for irrigation channels, irrigation losses, and the interaction between surface water with aquifers. The surface water network also includes major artificial irrigation canals. Among them the Cavour Canal, the Emiliano-Romagnolo Canal (CER) and the Muggia Canal are of the most important in terms of water flow derived from the natural network.

Figure 2 Po river basin major surface water bodies

2. Drought events 2003, 2006/07

In the recent past in Italy, the worst drought events occurred in 2003 and 2007. In 2003 very low total precipitation and high temperatures led to the lowest river discharges at the Pontelagoscuro measuring station (-6.99 m or 270 m³/s). In July 2003 the emergency situation for drought was declared in the regions Veneto, Lombardia, Piemonte, Emilia-Romagna and others across the country. Again in 2006-2007 the Po river basin experienced another severe drought. From October 2006 the whole Northern Italy was affected by anomalies in terms of precipitation and seasonal temperature. At the end of 2006 the
rainfall deficit reached 200mm and continued to rise until April/May. In May 2007, river discharges reached new lowest levels, lower than those of the 2003 and 2006.

ARPA Emilia Romagna (ARPA-ER) reported evidence of climate change effects on the water budget of its region. Even if this evidence is characterize by very high uncertainty it is worth to report that ARPA-ER recorded that Emilia Romagna mean temperature increased by about 2°C (~0.5°C/10y) over the past forty years (Cacciamani et al. 2010a) and precipitation decreased by some 20% (Cacciamani et al. 2008).

![Figure 3 - Trends in mean annual temperature (left) and precipitation (right) anomalies in Emilia Romagna in the last 40 years (1961-2008). The anomaly is calculated compared to reference climate of 1961-1990. Source: (Cacciamani, 2010).](image)

ARPA-ER also reported that the intensity of single rainfall events increased whereas the number of the rainfall events decreased. With almost 50% decrease from the previous long-term average, the decline in average precipitation is particularly pronounced in spring and summer periods. Snow cover and volume of glaciers show similar trends, as a consequence of shorter snow accumulation seasons. Cacciamani et al. (2010b) stated that the precipitation drop started in the early ‘80s. The difference in mean rainfall quantities in the last 25 years is estimated to be about 100 mm. It is a noticeable measure, corresponding to 10% of the mean annual rain in Emilia Romagna. However, these conclusions disagree
with the analysis complied by CNR (Nanni et al. 2007). This research focus on secular time series measured in 100 Italian stations. The decrease in precipitation is still visible, but it is defined as “minor and statistically little significant”.

Drought events are often coupled with high temperatures. In summer 2003, the average temperature amounted to 28.6°C, or some 5°C above the 1969-98 average. The precipitations fell down to 73 mm compared to average of 140.7 mm. In July (June and August respectively) the precipitation accounted only to 1.7 mm (31.5 mm, 39.7 mm). In total, the year witnessed precipitation 315.5 mm in Emilia, 60 mm less than average. However, this occurred after a very wet 2002, with more than 1000 mm (Lombroso L. et Quattrocchi S. 2003).

The main difference between 2003 and 2006/7 events is represented by the winter precipitations. In 2003 event was limited in the spring/summer period, and especially in June and July (ARPA-ER, 2003). The precipitations abundance in fall/winter 2003 allowed the complete recovering from the drought, reducing the consequences in the long run. The particular characteristics of 2006/7 event instead is represented by the scarce precipitations registered during the winter between the two years. Low winter precipitations strongly impact the quantity of water stored naturally, in the glaciers, or artificially, in the hydropower reservoirs of the alpine area, and consequently reduced the possibilities to mitigate the summer drought.

In 2007, the Po river basin underwent a strong water crisis. From January 2007 Northern Italy experienced considerable anomalies in terms of precipitation and seasonal temperature. The whole northern part of Italy had scarce precipitation (200 mm deficit at the end of 2006 in the plains and 30% deficit by April, despite the first three months of the year presenting average rainfall) and abrupt temperature rise (up to 3°C higher than average in December 2006 and to 6°C higher in January 2007). The snow-storages in the
Alpine Arc, settled in March, experienced a rapid melting due to the sudden temperature rise in April (Riva A. et Cucca R. 2007). Grates lakes of Lombardy were severely hit by the drought, and their levels were recorded several metres below average water levels. By May, the Po river discharge level was lower than values registered in 2003 and in 2006.

![Figure 4. Po River flow at Pontelagoscuro (cubic metres per second) years 2003, 2006 and 2007. Source: our elaboration based on ARPA Emilia Romagna Data.](image)

3. The Reno River and the Emilio Romagnolo Canal

The Emiliano Romagnolo Canal (CER) is one of the major artificial canals in the Po river basin. Even though the CER is not completely contained within the basin surface, it provides water to one of the most rural developed area of Italy and it is of great interest for understanding basin water buffer capacity. The project to build a canal aimed to supply fresh water to the eastern part of Romagna was designed since 1620. The construction of the actual canal started in 1955 with an estimated discounted cost of 550
millions €. It is one of the most important hydraulic infrastructures in Italy and it serves one of the most productive agricultural areas in Europe. CER runs for around 150 km, from the Po river near Ferrara to the Uso River near Rimini. Water delivered to the basin is around 7 million m³ per year. Several provinces benefit of its water: Ferrara, Bologna, Ravenna, Forlì-Cesena and Rimini. The water delivered is mainly used for irrigation and also for industrial purposes, drinking, leisure and water supply, e.g. the city of Ravenna withdraw its water from CER. The maximum water flow capacity is 184 m³/s and the total power used by the infrastructure is 14,858 kW. The administration of canal is managed by the CER Consortium, which aggregates several land reclamation and irrigation boards.

Figure 5 – Emiliano Romagnolo Canal. Source: Wikipedia.

CER water flow records over the last years are summarized in the following table:

<table>
<thead>
<tr>
<th>Water Volumes</th>
<th>2002</th>
<th>2003</th>
<th>2004</th>
<th>2005</th>
<th>2006</th>
<th>2007</th>
<th>2008</th>
<th>2009</th>
<th>2010</th>
</tr>
</thead>
<tbody>
<tr>
<td>Diverted by Po river at Palatone</td>
<td>176.814</td>
<td>321.009</td>
<td>239.059</td>
<td>230.906</td>
<td>244.633</td>
<td>290.497</td>
<td>228.963</td>
<td>230.978</td>
<td>176.748</td>
</tr>
<tr>
<td>Risen at Crevenzosa</td>
<td>145.876</td>
<td>211.733</td>
<td>164.937</td>
<td>164.354</td>
<td>192.993</td>
<td>258.647</td>
<td>199.416</td>
<td>196.092</td>
<td>155.642</td>
</tr>
<tr>
<td>At Sillano station</td>
<td>109.414</td>
<td>147.389</td>
<td>117.392</td>
<td>103.974</td>
<td>113.561</td>
<td>177.530</td>
<td>149.602</td>
<td>149.372</td>
<td>114.673</td>
</tr>
<tr>
<td>At Lamone station</td>
<td>21.209</td>
<td>43.970</td>
<td>36.909</td>
<td>34.428</td>
<td>37.326</td>
<td>46.693</td>
<td>48.654</td>
<td>52.238</td>
<td>41.618</td>
</tr>
</tbody>
</table>

Table 1 - CER water flow records. Source: CER, 2011
It is interesting to highlight that during the years of drought considered in this document (2003 and 2007), the canal registered a more intensive discharge from Po river in order to compensate the scarcity of water in the agricultural district of Romagna.

The Reno river is the major river in Emilia Romagna region. It runs over 212 km with an average water flow of 95 m3/s. It is the tenth Italian river as for length and basin surface. The original course of the river was flowing towards Ferrara causing major floods in the low lands, and then leading and entering into the Po river. Because of these inundations, the river was diverted in the 18th century into an artificial channel, the so called Cavo Benedettino, which avoided the connection to the Po river and the recurrent floods. Nowadays, the Reno river is the only major river of Emilia Romagna which is not affluent of the Po river and flows directly into the Adriatic sea. Its basin has a population of two million inhabitants (Pistoia, Prato, Firenze, Bologna, Modena, Ferrara, Ravenna) and it includes major industrial and highly productive agriculture areas.
4. Drought event 2011/12

The last drought event occurred in the Po river basin is here described. Even though the Ridracoli Dam, which is located near Forli, is outside of the borders of the basin, the problem of water supply to the Romagna region is strictly connected to the Po river, because of the water transfer provided by the Emiliano Romagnolo Canal (CER) and of its proximity to the basin. The Ridracoli dam was built between 1974 and 1982 to create a reservoir of high quality water in the middle of Emilia Romagna Region, on the Appennini mountain chain. The original aim was to deliver high quality drinkable water to the drought prone area of the province Forli-Cesena. In the last decades the reservoirs has been largely overused, the water has been supplied to almost the entire Romagna region: Ravenna, Faenza, Forli, Cesena, Rimini and San Marino.

The maximum volume storable is about 33 million m³, the reservoir supplies drinkable water to a large part of the area, with more of 1,000,000 inhabitants. Due to the concentration of pollutant, the water of the Ridracoli lake cannot be supplied for domestic
consumption when the level of the water falls under the threshold of 5 million m³. The reservoir is strongly impacted by the rainfall cycle and, looking at the historical data, the most critical period is represented by the months of October and November. In this period the water scarcity brings the lake to its annual minimum level and, in years in which drought conditions affect the water availability, the probability to pass the critical threshold is very high.

Drought events registered in 2003 and 2006/2007 impacted in a different way on the Ridracoli reservoir. The differences are mainly represented by the different precipitation levels registered during the winter and spring time and, consequently, by the level of the water stored at the beginning of the drought event. The most critical situation has been registered in 2007. The drought registered in 2006 and the scarce precipitations during the winter did not allow to reach the usual maximum storage on May. In this context, the strong drought of 2007 brought the reservoir almost at the crucial level of 5 million m³.

Figure 7 Water volumes (mil m3) stored in the Reservoir Ridracoli over the period 2003-2007, intra-annual variability. Source: own elaboration based on Diga Ridracoli data.
The critical level, almost touched in the previous drought events, has been reached on November 2011, peak of the emergency situation of water scarcity during the water crisis registered on fall 2011.

Fall and winter seasons of the period across 2011/2012 have been characterised by a strong water scarcity condition. The precipitation shortage, the small amount of snow, and the scarce quantity of water stored in the mountainous reservoirs seriously threatened the normal water use, even for domestic purposes in the north-eastern area of the Country. The zones more impacted by the 2011 winter drought were the regions Emilia Romagna and Veneto. The administrations of the two Regions declared the official “State of Emergency” respectively on December 2011 (Emilia Romagna) and April 2012 (Veneto).
Regarding Emilia Romagna, the most critical area has been represented by the locations where water supply is ensured by the Ridracoli reservoir. The regional authority imposed extraordinary measures of water restriction for the entire Romagna. The local ATOs (Rimini, Ravenna and Forlì/Cesena) were requested to immediately localise alternative possible water sources. Moreover the alternative sources exploitation rate, regulated by concessions, has been consistently increased. The water provision for non-vital uses has been consistently reduced, except for the fire control sprinkler systems (Ordinanza del Presidente della Giunta regionale n. 177 del 07 Dicembre 2011).

The regional administration of Veneto temporarily infringed the regulations about the environmental flow of the main rivers. The water quantity authorised for irrigation purposes have been reduced by 40 per cent. The environmental flow of the Adige and the Po river has been fixed respectively to 80 and 450 m³/sec, in order to contrast the salt water intrusion. All water non-basic uses (car washing, fountains, garden irrigation, etc.) have been restricted (Ordinanza del Presidente della Giunta regionale n. 67 del 03 Aprile 2012 e n. 84 del 2 Maggio 2012).

In both the cases, the State of Emergency has been put in practice till the end of May of 2012. Precipitation rate turned to normal conditions in the period between April and May 2012. The Ridracoli reservoir reached again its maximum level in May 7th 2012 with 33 million cubic meters of water.

References
Research Task 4.2 Output 11

Analysis of asymmetric information in the Tagus River Basin

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Authors  Davide Viaggi
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4. Issues to be considered in empirical analysis ............................................................................................. 7
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1. Objectives

The objective of this preliminary note is to provide some background to the treatment of asymmetric information (AI) in the scarcity group of EPIWATER. It particularly focuses on definitions and issues to be accounted for to deal with asymmetric information in the empirical analysis of water markets. It particularly supports the Tagus river basin case study.

2. Definitions

Information is a basic issue in planning, economic activities and public policy. In economic literature AI is studied in connection to the effects it has on transaction among actors, being them market actors or a regulator and regulated agent. AI has been studied in a number of fields, including environmental regulation (Salanié 1998; Laffont and Martimort 2002; Macho-Stadler and Pérez-Castrillo 1997). The design of a problem under AI distinguishes usually two actors. The principal and the agent. The principal in interested in the outcome of some activity by the agent. The related literature distinguishes two main asymmetric information situations. The first, adverse selection, is the case in which there are different agent types, but the agent type is unknown to the principal. The second, moral hazard, is the case in which an action by the agent is unobservable by the principal. Both situations can occur simultaneously. Literature about AI concern the study of how AI can affect the transaction, including the situation in which the willingness to participate in the transaction collapses to zero due to the inability of the principal to predict/control the outcome. Also, the literature studies the way contracts and institutional settings may be designed in order to overcome (usually at a cost) the AI problem. For example, under adverse selection conditions, contracts may be designed in order to provide incentives to the agents to reveal their characteristics by choosing one among a menu a contracts. In case of moral hazard, different contracts may be designed relating a
payment to an observable proxy that is probabilistically related to the unobserved action, in order to provide incentives to the agent to perform the action as sought by the principal (Laffont and Martimort 2002).

3. Literature related to water

Information is well known as a problem in water management. This usually relates to availability of reliable data about water flows, and in particular on individual uses. The impact of such information deficiencies impacts water management at different levels. The most commonly cited ones are the poor planning of water resource use due to insufficient information and the difficulties for regulation in case of information asymmetries.

The former is a straightforward issue based on the notion that without sufficient information policy provisions will be sub-optimal, or, in real life, yield unexpected results. The lack of sufficient information for supporting public policies in the water sector is an issue often recalled in the literature and policy documents (Chambers and Trengove, 2009; OECD, 2010). The recent blueprint on water (European Commission, 2012) emphasises the need for an information basis in water policy and highlights the achievements obtained in the EU about this issue.

The latter issue is treated in the water pricing literature. Information asymmetries are recalled as a difficulty in implementing water pricing systems (Johansson, 2000; Johansson et al., 2002; Tsur, 2009; Mohayidin et al., 2009) and becomes particularly difficulty when a regulator intends to implement volumetric water pricing where water metering is not possible (or not in place anyway).

In spite of the relevance of the problem of information asymmetries for water management in agriculture, up until now research has paid very little attention to formally studying this issue. Most of the paper concern water regulation and payments, rather than
markets, i.e. study the way a regulator may design contracts to deal with incentive pricing under conditions of unmetered water. Smith and Tsur (1997) and Tsur (2000) were first to explore this issue by providing a classification of cases in which asymmetric information leads to the need for the mechanism design approach (i.e. menus of contracts) to deal with information asymmetries in water pricing. With regard to the EU context, Bazzani et al. (2004) use the structure of a Principal-Agent model to quantify water demand and optimal regulation from the policy maker’s point of view, when implementing both Full Cost Recovery and the Polluter Pays Principle and when monitoring costs and sanction options are considered. Gallerani et al. (2005) develop this approach through a Principal-Agent model under asymmetric information and adverse selection, searching for optimal water contracts under the constraints set by the WFD, referring in particular to individual cost recovery constraints, and transaction costs related to payments. Viaggi et al. (2010) develop the same framework in the direction of extending the model to more than two farm types, and considering an aggregate cost recovery constraint. In addition, they use the model to simulate how different market scenarios for agricultural products would affect optimal contract design.

The study of AI in connection to water-related markets is rather underdeveloped in the literature and tend to concern very specific settings.

With respect to water markets, there are two sides of the AI issue.

First, water markets can be seen as an instrument to solve AI issues. Delegating water allocation to a decentralised mechanism, AI between the agent and a regulator seeking optimal water allocation can potentially be solved (e.g. the regulator does no need to know quantities and marginal values of water for individual agents, as they express it through the market) (Tsur (2009)). In this perspective, AI can be relevant in evaluating the performance of markets with respect to alternative real life policies.
On the other hand, AI can affect the functioning of the market itself, when they occur among agents (and hence affect the transactions) or when the affect the initial allocation of property rights.

The only paper directly addressing the connection between AI and quantitative water trading is Dridi and Khanna (2005). They develop a basic model of quota and fee setting for the provision of water to farmers by regulators. They further use this model to discuss how AI affects water trading and technology adoption, and provide a numerical illustration for southern California and Arizona. The main conclusion is that even under asymmetric information, a secondary market improves the allocation of water resources and induces additional adoption of modern irrigation technologies.

AI in relation to pollutants trading has been treated by Nguyen et al. (2013), that develop a stochastic agent-based model to simulate water quality trading with asymmetric information, uncertainty and transaction costs.

4. Issues to be considered in empirical analysis

To deal with the issue of AI and water markets a primary issue would be to provide a systematic classification of the areas of the systems in which AI can emerge, e.g. allocation of rights, individual transaction, ex post enforcement, etc. (as long as we know this is not available in the literature). This could have a key role in a meaningful assessment of the performance of water markets as compared to other EPI.

Secondly, in order to incorporate asymmetric information into empirical analysis of water markets, the following issues should be considered:

1. Is there heterogeneity in the actors playing on the market (seller, buyers) in terms of willingness to pay/accept for water?
2. Are there opportunities for non-observable actions buy actors?
3. Is there lack of easily observable precise proxies for unobserved characteristics/actions?

In case of 1 and 3 there is potential for adverse selection. In case of 2 and 3 of moral hazard. In case 1, 2, and 3 both.

In case Asymmetric information is relevant, there are three main lines of study to address it.

First of all, decision can be simply studied considering asymmetric information. Under asymmetric information decisions by actors will be different and the outcome in terms of policy evaluation parameters will be also different.

Secondly, means to generate the necessary information may be studied, e.g. water metering, monitoring, self reporting, public information, etc.

Third (alone or in combination with the above) contractual arrangements needed to deal with the information problem can be designed using a mechanism design approach. The latter is feasible when a proxy exist which is probabilistically related to either the characteristics of the agent (adverse selection) or the sought outcome (moral hazard).
References


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Research Task 4.2 Output 12

Stakeholder engagement

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Authors: Simon McCarthy
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1. Why stakeholder engagement?

Whilst the involvement of stakeholders in environmental decision making is not entirely new it is their increased diversity and the strength and new types of roles of partnership working that has changed (Walker et al. 2010). There is a policy movement at international and regional levels towards individual and community participation in decision making for example the Aarhus Convention (UNECE 1998), EU Water Framework Directive (2000), UN Hyogo Framework for Action (HFA) (2005) and in disaster management the EU Floods Directive (2007).

It is increasingly recognised that the differences in policy intention and actual policy implementation is often informed by the action and inaction of diverse sets of stakeholders, the networks and partnerships involved and ‘multi-level governance’ (Walker et al. 2010). It is when stakeholder participation is required in policy implementation but genuine participation (Arnstein 1969) does not take place that problems arise resulting in mistrust and disillusionment (Bickerstaff and Walker 2005, Petts 2006). Understanding the points of view, needs and behaviours of those stakeholders that will potentially be affected or affect the policy implementation is therefore critical to affect meaningful change.

2. The approach

It is important to remind ourselves of what we wish to achieve in this work package. Ideally, to understand the impacts of the EPI from different stakeholder perspectives it would be best to assess the initial conditions prior to the EPI’s application. These conditions should then be compared with the resulting state after the application of the EPI to assess the impact or contribution of the EPI (See Figure B-2 below).
However, the reality is that many EPI’s are implemented over many years and may have started before this project and so the ex-post and ex-ante approach was developed. Having now undertaken the ex-post exercise in work package 3 an ex-ante exercise will be undertaken in this work package. This involves making judgements on how an EPI might be expected to change the impact areas, based on pre-determined scenarios.

To capture all of the relevant information for each case study, the methodology comprises two essential components:

1) Analysis of secondary data (i.e. case study reports and surveys)

2) Interviews with stakeholders to verify secondary data and collect the enhanced measurement information.

It is the interpretation of these **two sources** of information **together** that should be presented in the results.

---

Figure B-2. Assessment Framework for Task 2.3
3. Case Study Design

The issues surrounding this case are complex but in order to clearly measure how the EPI might inform change the case study needs to be focused and well defined in relation to the EPI.

3.1. What are we measuring?

Measurement will be a combination of financial and social indicators in response to the current situation and possible future situations communicated as scenarios. For social indicators we refer to the opinions and perceptions of stakeholders.

The EPI is water trading and there are three scenarios where measurement is required in the case:

1. Water scarcity (current long term situation)
2. Emergency with insurance compensation (current short term situation)
3. Emergency with trading and/or insurance compensation (new scenario)

The focus for analysis is on water trading whilst recognising the important role compensation plays in an emergency declared drought.

3.2. Where are we measuring?

The Tagus – Segura case could involve both inter and intra-basin applications of the EPI with the intra-basin within Segura. It has been decided that the intra-basin trading is currently an unrealistic scenario (25/04/2012 first round stakeholder report). The inter-basin trading has already been tested on a small scale and appears to function. In order to keep the research focused and to concentrate resources the inter-basin scenario will be used. Findings from the initial small scale inter-basin trading will inform improvements to
the new EPI described in the scenario but also stakeholders will take part in the scenario testing based on their previous real life experience.

The case study will include areas that are the least (or even not) dependent on ground water sources.

If required by the research a separate sample of stakeholders where ground water might be an issue but trading is still a viable option will be included such as the areas of ....

3.3. Who are we measuring?

Water scarcity is the normal condition for both the case study areas. There is a sequence and different scales of interaction in the announcement of an emergency and the possible instigation of water trading. Put very simply first the representatives of the farmers at a national level and also at the basin level in Tagus and Segura have to agree that water transfer will be allowed through a decree. Then decisions can be made by the representatives of the big farms and farmers who will give water in Tagus and who will buy water in Segura.

Ideally in order to obtain a clear understanding of the stakeholder context for the EPI three different types of stakeholders will be interviewed:

a. Those who will be directly involved in the trading decisions eg: .... organisations from previous stakeholder interaction plus any new ones ....

b. Those who are directly affected by those decisions eg: third parties (environment, tourism, utilities, water users ....

c. Those who have an influence over the policy eg: academic, Government minister etc
However, resources and opportunities might dictate that just a. and b. are focused on. It has been decided that national, regional and smaller scale farmers’ representatives will be the key respondents. It is felt that they are good representatives of farmers but perhaps biased towards the larger farms. Towards the end of the fieldwork a sample of Segura farmers will be interviewed in order to gain a better understanding of the issues informing the decisions they make. This will also help clarify the representativeness of the findings from those representing the farmers’ decisions.

For both the representatives and the farmers a spread of characteristics will be sought. For farmers this might be crop type / farm size / location / water rights. For ‘representatives’ perhaps in terms of the farmers they represent and the constraints the ‘representatives’ work under. The number of ‘representatives’ available for interview may be more restricted than farmers.

In all the designs the types of trading decisions will need to be presented either as realistic to the basin they occupy or that realism represented in the scenarios. Three groups that need to be represented as decisions:

1. Buyers (~ Segura area)
2. Sellers (~ Tagus area)
3. Non-participants who decide not to be involved.

The last group could be identified from the small scale trading that took place and interviewed. If not respondents can instead be asked in the interview theoretically if they would or would not participate and why.

In order to help bring the scenarios to life it is important that those interviewed must have experienced the last emergency (2006) while doing their current job.
3.4. **When will we measure?**

Interviews will be focused around three stakeholder meetings each lasting two days. It will be important that the interviews are scheduled so that respondents can be interviewed individually in private as well as monitoring their group discussions around the issues at the meetings.

The scenarios are critical and so time should be allocated well before the meetings to carefully develop the scenario descriptions. They will need to be as realistic as possible so that the stakeholders can bring them to life through their own interpretations and experience.

The three meetings are:

2. February 2013, Madrid

It is anticipated that there will be a significant number of stakeholders at each meeting representing a range of characteristics for the regions. These will be noted for the research.

2.5. **How will we measure?**

The objective is to gain an understanding of the decision processes and challenges of stakeholders’ current scarcity and emergency decision making and impacts as well as how the EPI might affect the stakeholders in the future through the scenarios. It will indicate if for this case study the EPI will be acceptable but more importantly reveal the underlying processes to inform the transferability of the EPI to other cases. For this reason a qualitative approach has been adopted.

The capabilities and understanding of the respondents will greatly influence the measurement approach hence the importance of clarifying the case study and possible stakeholders in advance.
The interview discussion could take four stages:

1. An introduction and understanding of respondent background
2. Exploration of the current context and both financial and social indicators
3. Reaction to the scenarios measuring social indicators as per the enhanced measurement approach
4. Respondent’s reaction to the interviewers summary interpretation of the whole interview

A full interview guide will be required. However, at this stage it can be said the social indicator measurement approach will be based on the ex-post enhanced measurement but this time the scenario descriptions are key. Again the interviews would take the form of one to one structured discussion with the respondent probably lasting an hour. The length of the interviews will depend on the number of scenarios which are in turn dependent on the case study design.

At this point we can say it would be a sequential approach in stage 3 using the interview grid to the following scenario types:

A. Water scarcity (current situation)

B1. Emergency with insurance compensation (current situation)

B2. Emergency with trading and/or insurance compensation (new scenario)
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Scenario A would measure their current rating on a scale (perhaps a 5 point scale) of each impact category in their lives or organisation. Also the respondent will be asked which are for them the most important categories. Then the respondent will be asked in turn about scenarios B1 and then B2 if or how each category might change from the first rating.
Depending on the case study design a respondent might only ever be a buyer or a seller but if they could be both then the B2 scenario will need to take this into account and the respondent asked from each trading perspective in turn.

This is just an outline of the issues and the style of the measurement could change depending on the case study design which is finally agreed.

References


Research Task 4.2 Output 13
EPI4Drought: an agent based model for assessing water trading between Spanish Irrigation Communities during a drought event

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Filename | EP4Drought: an agent based model for assessing water trading between Spanish Irrigation Communities during a drought event
Authors | Christophe Viavattene (FHRC – Middlesex University)
Carlos Dionisio Pérez Blanco (IMDEA Water)
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1. Introduction

Water scarcity is the most pressing environmental issue in many southern European regions. This situation is to a great extent attributable to agriculture. Consequently, policy makers in drought prone areas have called for measures in this sector to alleviate water scarcity. These measures have consisted so far on supply oriented policies, such as the construction of major infrastructures or the modernization of irrigation devices, and on the enactment of restrictive laws, whereas water demand policies have been excluded from the policy mix. Paradoxically, this scheme has ended up increasing water demand, reducing water availability and undermining the robustness and resiliency of the system and its ability to cope with future droughts (Gómez and Pérez, 2012).

More recently the literature has encouraged the implementation of economic instruments and in particular of water markets as an inexpensive and efficient way of reallocating water among users (Ranjan and Shogren, 2006; Cave, 2009; Rey et al., 2011). Actually, the economic and environmental outcomes of this instrument differ a lot from one region to other: while the Australian and Chilean experiences indicate that a market based on nominal rights and transfers may not be sustainable or equitable, the principle of effective use characteristic of the US (and the EU) in which water remains a public resource subject to forfeiture if not used may provide under certain preconditions a Pareto improvement.

In the frame of the EPI-Water project, MU and IMDEA have designed an Agent Based Model (ABM) to assess the potential of water markets to attain a better allocation in the particular case of the Tagus and Segura interconnected river basins in Central and South-Eastern Spain. Agent-based simulation aims at portraying social entities’, behaviour and relationship in order to study the global behaviour of their population and to simulate emerging organisations. It is based on a bottom-up rule-based mechanism. A multi-agent system is typically composed of an environment (a space), a set of situated objects, an assembly of agents (active entities), a number of relations between different objects, a set of agents’ capacities (perceiving, communicating, behaviour, interaction with other objects…) and a set of defined rules (universe laws) (Ferber, 1997). ABM results from the convergence of two aspects. The first aspect is the idea that the behaviour of large groups can be understood on the basis of very simple interaction rules, so that individuals act essentially as automata responding to a few key stimuli in their environment (Ball, 2004). The second aspect is the past development in DIA (Distributed Intelligence Artificial), of which the objective is to reproduce the knowledge and reasoning of several heterogeneous agents that need to coordinate to jointly solve planning problems (Bousquet and Le Page, 2004). ABM has already been successfully used in various contexts from physical modelling, ecological modelling and social behaviour modelling to more complex modelling such as CHAN (Coupled Human And
Natural systems) or environmental modelling (Tweedale et al., 2007; An, 2001; Bithell et al., 2008).

In this research project the economic model, called EPI4Drought, (Figure 1) has been developed on the netlogo platform (http://ccl.northwestern.edu/netlogo/). The software is free of charge and open-source. This report presents the modelling context (i.e. the Tagus-Segura Inter-basin water trading scheme), the model principles and rules and the results obtained under different type of trading schemes.

![Figure 1: The EPI4Drought ABM interface](image)

2. The Tagus-Segura Inter-basin water trading scheme

Water demand in the Segura River Basin (SRB) amounts to 1 900 million m³ per annum while average renewable resources are estimated to be only 760 million cubic meters per annum, making the SRB the most overexploited river basin in Europe (EEA, 2009). Shortage of renewable resources is partially compensated by an inter-basin water transfer from the Tagus River Basin (TRB) (circa 330 million m³/year)
that, nevertheless, since its opening in 1985 has been always below the maximum capacity of the transfer of 600 million m³/year. The resulting deficit is mostly covered by the overexploitation of aquifers and has resulted in a significant environmental deterioration (SRBA, 2008).

In contrast to that, water demand in the Tagus River Basin (TRB) amounts to 2 600 million m³ over an average resource availability of 12 000 million m³. The TRB is the largest of the Iberian Peninsula (its Spanish section covers 55 750 km²), and despite momentary local scarcity problems and the high variability in water resources, drought vulnerability is still moderate in the river basin (TRBA, 2010).

![Figure 2: Tagus (Orange) and Segura (Green) Interconnected River Basins](#)

The informal nature of an increasing share of water abstractions in the SRB, especially during drought events, is both the result of the water scarcity and the socioeconomic relevance of irrigation for an area which has one of the most productive agricultural sectors in Europe, on which the economy relies strongly (Pérez et al., 2011). The Segura River Basin Authority has recognized the importance of this sector and the need to guarantee its viability in the years to come, thus transforming the informal rights into de facto rights (SRBA, 1998; SRBA, 2008; SRBA, 2010). Contrary to this, after the approval of the EU Water Framework Directive (EC, 2000) a more prominent role has been given to the environmental and urban uses to the detriment of other productive uses, including agriculture, the basin’s most relevant water user (85% of total water demand) (SRBA, 2008). For example, the recently approved Drought Management Plans (DMPs) (SRBA, 2010; TRBA, 2007) introduce clear restrictions to water allocation in agriculture during drought events.

---

1The actual capacity of the TSWT is 1 000 million cubic meters per year, but it has been limited to 600 million cubic meters per year by law (SRBA, 2013).
The new regulatory framework also makes possible the implementation of economic policy instruments with the potential to meet both the economic and environmental objectives at stake, such as water markets (EC, 2000; Water Act 29/1985, Act 46/1999, RD 1/2001, Water Act 63/2003).

In the new Water Law two main different procedures for inter-basin water trading were allowed for, both requiring approval from the corresponding River Basin Authority (Calatrava and Garrido, 2005):

Lease Contracts (“Contratos de cesión”): Direct trading among concession holders who privately agree on the conditions for the temporary lease of public water concessions;

Water Banks (“Centros de intercambio”) that are publicly-run water banks: Aim to speed water transfers during periods of scarcity and to disseminate information about volumes exchanged and prices paid.

Water use tradable rights in Spain are only granted when a set of prerequisites are met (see Rey et al., 2011). In the case of inter-basin water markets using previously existing infrastructures, such as the Tagus-Segura water transfer, a specific legal framework has to be in place in the form of a Royal Decree (RD 15/2005).

The inter-regional conflicts arising from water transfers in Spain have been so far an important limit for the development of these markets, with Royal Decrees being approved only under emergency junctures. In spite of this barrier, inter-basin water markets have been the most successful reallocation instrument in terms of volume traded (Rey et al., 2011). For example, water trade from the TRB to the SRB only in 2006 surpassed that of all the water exchanges approved before within the SRB, when farmers from the Upper Tagus Basin (Comunidad de Regantes de Estremera) agreed to transfer 31.5 million cubic meters annually during three years to farmers in the Segura Basin through a particular type of lease contract supervised by the authorities (though this may be arguable, since illegal intra-basin water transfers are not accounted for in the statistics; see for example Hernández-Mora and De Stefano, 2013).

The Agricultural Demand Units

The model aims at representing the behaviour of the irrigation communities. But as no information is available at such scale, the UDAs (Unidades de Demanda Agraria or Agricultural Demand Units) have been used as representing of a group of irrigation communities (Figure 3). Given the high complexity of the political framework surrounding the Tagus-Segura water transfer, only the UDAs in the TRB headwaters (8 UDAs) and the UDAs of the SRB (21 UDAs) which have formal water allocations over the resources from the transfer are represented in the model.
For each UDA information may be obtained on the percentage of area for different crops categories, the water requirement for each crop category and the irrigation efficiency and price (TRBA, 2013; SRBA, 2013). The crops categories considered in the model are: almond tree, winter cereals, spring cereals, summer cereals, citrus trees, industrial crops, fruit tree (stone-pit), fruit tree (pip), horticulture (bulb), horticulture (cauliflower, artichoke), horticulture (fruit), horticulture (leaf), horticulture (green house), horticulture (root), horticulture (tubercular), leguminous plant, olive grove, vineyard (fruit), vineyard (wine). However the database does not provide information on the irrigated net marginal value and the rain fed net marginal value. Average values were obtained from regional statistics (MARM, 2012) by averaging the associated parameters of specific crops into UDAs group categories. The parameters considered are the yield value, direct costs, machinery cost, labour cost, water costs and subsidies. A white noise is also estimated to take into account any source of variability in the final yield value (excluding droughts²).

---
² Therefore, this white noise was obtained from the years where no drought was declared.
3. Model Principles

The aim of the model is to assess if irrigation communities from different catchments may benefit from inter-basin water trading in a drought situation. The model has been developed to fit the Tagus-Segura River Basin case study. However, the theoretical framework of the model, the methodology and the expected outcomes could be used on other catchments with similar information available.

3.1. The Environment: Drought Stochastic approach

The model simulates yearly events independently. Each early event represents the same year but with different conditions. No learning process or no past experience is therefore considered in the current model. The trigger effect in the model is the announcement of a drought. In Drought Management Plans (DMPs), four levels of severities are considered per catchment: normality, pre-alert, alert and emergency (TRBA, 2007; SRBA, 2008). For each level of severity the level of water allocation is reduced. In a normal situation it is considered that the agronomic water requirements (i.e. evapotranspiration) are satisfied. The methodology used at this stage is based on the work by Gómez and Pérez (2012).

The methodology adjusts a Probability Density Function (PDF) for the relevant variables in the drought index of the sub-basins implied in inter-basin trade. This way the probability of every possible drought event is obtained using two types of functions: the Gamma function is used to adjust the rainfall, runoff and the piezometric level PDFs (Gómez and Pérez, 2012; Pérez et al., 2011; Martin et al., 2001; McWorther et al., 1966), while the Weibull function is used for the water stock in reservoirs (Gómez-Ramos et al., 2002).

The Gamma PDF is a function of a scale $a$ and a shape $b$ parameters and ascribes a probability $p_i$ $(i = 1,...,3)$ to every value of the variable $x_i$ $(i = 1,...,3)$:

$$p_i = z(x_i | a, b) = \frac{1}{b^a \Gamma(a)} x_i^{a-1} \exp(-\frac{x_i}{b})$$

where $x_1$ stands for the rainfall, $x_2$ for the piezometric levels and $x_3$ for the runoff. $p_1$, $p_2$ and $p_3$ are the corresponding probabilities.

The Weibull PDF is a function of a scale $c$ and a shape $d$ parameters. The Weibull PDF ascribes a probability $(p_4)$ to every value of the water stock in reservoirs $(x_4)$:
\[ p_4 = j(x_4|c,d) = \frac{d}{c} \left( \frac{c}{d} \right)^{d-1} \exp\left( - \left( \frac{x_4}{c} \right)^d \right) \]

From the PDF’s estimated parameters, the value and the likelihood of every drought index, as well as its associated water restrictions, can be estimated. Drought indexes can be obtained from a single variable or from the combination of up to four of them, weighted by a coefficient predetermined in the corresponding DMP, \( b_i \). The drought index is thus obtained as follows:

\[ I_e = \sum_{i=1}^{4} b_i \cdot I_e,x_i \]

with:

\[ I_e,x_i = \begin{cases} \frac{x_{ij} - x_{i_{\text{min}}}}{2(x_{i_{\text{med}}} - x_{i_{\text{min}}})}, & \text{if } x_{ij} < x_{i_{\text{med}}} \\ \frac{1}{2} \left[ 1 + \frac{x_{ij} - x_{i_{\text{med}}}}{x_{i_{\text{max}}} - x_{i_{\text{med}}}} \right], & \text{if } x_{ij} \geq x_{i_{\text{med}}} \end{cases} \]

where \( x_{ij} \) is the observed value, and \( x_{i_{\text{med}}}, x_{i_{\text{max}}} \) and \( x_{i_{\text{min}}} \) are the average, maximum and minimum historic values, respectively.

Finally, the ascribed probability is obtained as:

\[ p_e = \prod_{i=1}^{4} h(p_i) \]

where:

\[ h(p_i) = \begin{cases} 1, & \text{if } b_i = 0 \\ p_i, & \text{if } b_i > 0 \end{cases} \]

It is also possible to obtain the probability of every drought stage in both the Segura and Tagus River Basins by aggregating the probability of all the drought indexes that fall within each threshold. The following dummy variables are defined:
\[
N_{I_e} = \begin{cases} 1, & \text{if } I_e > I_{e,e} \\ 0, & \text{if } I_e \leq I_{e,e} \end{cases}
\]

\[
P_{I_e} = \begin{cases} 1, & \text{if } I_{e,a} < I_e \leq I_{e,e} \\ 0, & \text{otherwise} \end{cases}
\]

\[
A_{I_e} = \begin{cases} 1, & \text{if } I_{e,a} < I_e \leq I_{e,e} \\ 0, & \text{otherwise} \end{cases}
\]

\[
E_{I_e} = \begin{cases} 1, & \text{if } I_e \leq I_{e,e} \\ 0, & \text{if } I_e > I_{e,e} \end{cases}
\]

where \( I_{e,x}, I_{e,a} \) and \( I_{e,e} \) are the pre-alert, alert and emergency thresholds, respectively.

The probability of every drought threshold (normality, \( q_N \); pre-alert, \( q_P \); alert, \( q_A \); and emergency, \( q_E \)) is obtained as follows:

\[
q_N = \int_{x_1=0}^{\max x_1} \int_{x_2=0}^{\max x_2} \int_{x_3=0}^{\max x_3} \int_{x_4=0}^{\max x_4} \left( N_{I_e} \prod_{i=1}^{4} h(p_i) \right)
\]

\[
q_P = \int_{x_1=0}^{\max x_1} \int_{x_2=0}^{\max x_2} \int_{x_3=0}^{\max x_3} \int_{x_4=0}^{\max x_4} \left( P_{I_e} \prod_{i=1}^{4} h(p_i) \right)
\]

\[
q_A = \int_{x_1=0}^{\max x_1} \int_{x_2=0}^{\max x_2} \int_{x_3=0}^{\max x_3} \int_{x_4=0}^{\max x_4} \left( A_{I_e} \prod_{i=1}^{4} h(p_i) \right)
\]

\[
q_E = \int_{x_1=0}^{\max x_1} \int_{x_2=0}^{\max x_2} \int_{x_3=0}^{\max x_3} \int_{x_4=0}^{\max x_4} \left( E_{I_e} \prod_{i=1}^{4} h(p_i) \right)
\]

where \( \max x_i \) is the value of the variable \( x_i \) that makes the cumulative density function equal to 1.

Finally, the relative water allocation (\( w_{I_e} \)) remains to be defined. During a normal hydrological year, all the agronomic water requirements are satisfied (i.e., water allocation equals the agronomic water requirements, \( w_{I_e} = W \)). In the event of a drought, DMPs define the water constraints that will come into force for every drought scenario (\( y \)). Therefore, the amount of water allocated (\( w_{I_e} \)) is obtained as follows:

\[
w_{I_e} = y \times W
\]
For example, in the SRB, water availability during an emergency is reduced by 50% \((y = 0.5)\), by 25% in the case of an alert \((y = 0.75)\) and by 10% in the case of a pre-alert \((y = 0.9)\) (SRBA, 2008).

### Table 1: Drought probability and associated water allocation rate

<table>
<thead>
<tr>
<th>Level</th>
<th>Probability ((q))</th>
<th>Water allocation rate ((y))</th>
<th>Probability ((q))</th>
<th>Water allocation rate ((y))</th>
</tr>
</thead>
<tbody>
<tr>
<td>Emergency (E)</td>
<td>0.18</td>
<td>0.5</td>
<td>0.19</td>
<td>0.5</td>
</tr>
<tr>
<td>Alert (A)</td>
<td>0.31</td>
<td>0.75</td>
<td>0.26</td>
<td>0.77</td>
</tr>
<tr>
<td>Pre-alert (P)</td>
<td>0.29</td>
<td>0.9</td>
<td>0.26</td>
<td>0.95</td>
</tr>
<tr>
<td>Normal (N)</td>
<td>0.21</td>
<td>1</td>
<td>0.29</td>
<td>1</td>
</tr>
</tbody>
</table>

### 3.2. The Agents

The model represents the behaviour of irrigation communities, the Unidades de Demanda Agraria (Agricultural Demand Units or UDAs), which is the basic irrigation unit with available data in Spain. The UDAs comprise exploitations sharing the source of their water resources, administrative characteristics, hydrological characteristics and/or a territory (SRBA, 1998; TRBA, 1998).

Each agent or UDA is in charge of allocating water to different types of crops in order to maximize their revenues. In most agricultural models, the agents pay attention to one of the following three variables: wealth, income and gain/loss (Hardaker et al., 2004). It could be expected though that the agents incorporate indirectly all these variables in their assessment, as gain/loss is a marginal change in wealth and wealth is the capitalized value of all the present and future incomes. In this approach we use the marginal change in wealth as the argument. Thus, the objective function of each UDA is obtained as follows:

\[
\text{Max } \Pi = \sum_{i=1}^{N} \left( P_i * r_i - \sum_{j=1}^{T} c_{ij} \right) * s_i * \Omega(w_i) + u_i + \left( P_i * r_{i,RF} - \sum_{j=1}^{T} c_{ij,RF} \right) * s_i * (1 - \Omega(w_i))
\]

Subject to:

\[
\sum_{i=1}^{N} w_i \leq w
\]
\[ w \geq w_{\text{min}} \times s_{\text{ligneous}} \]

where \( i = 1 \ldots N \) are the different crops in the UDA, \( \Pi \) is the revenue, \( P_i \) is the price of the crop in EUR per kg, \( r_i \) and \( r_i,RF \) are the average yield (in kg per hectare) for crop \( l \) under irrigation and rainfed agriculture\(^3\), respectively, \( c_{ij} \) and \( c_{ij,RF} \) are the costs involved in production (in EUR per hectare) under irrigation and rainfed agriculture, respectively, \( s_i \) is the surface allocated to crop \( i \) (in hectares), \( \Omega(w_i) \) represents the percentage of the surface of the crop \( i \) that can be irrigated (dependent on the total water allocation for that crop, \( w_i \)) and \( u_i \) and \( u_{i,RF} \) are white noises that capture any source of revenue variability apart from water scarcity, such as plagues, hail, floods, price volatility, etc., under irrigation and rainfed agriculture, respectively.

The UDA needs to consider that ligneous crops require a minimum amount of water per hectare in order to secure their survival (\( w_{\text{min}} \)); therefore, prior to the maximization process each agent delivers that minimum amount of water (\( w_{\text{min}} \times s_{\text{ligneous}} \)). Since the irrigation of ligneous crops is given a high priority by the law (SRBA, 2008; TRBA, 2007), water resources available need to be at least equal to \( w \geq w_{\text{min}} \times s_{\text{ligneous}} \). In any case, the amount of water applied (\( \sum_{i=1}^{N} w_i \)) cannot exceed the amount of water available (\( w \)), where \( w = w_{\text{le}} + m \) is the total amount of water available, being \( w_{\text{le}} \) the allocated water in the drought scenario (obtained in the previous section) and \( m \) the water purchased in the water market (in this baseline scenario, \( m = 0 \)).

During a drought event, agents will solve the problem above and irrigate those crops that maximize the objective function. With no market, the remaining crops will be left non-irrigated.

3.2.1. Agents under a water market

In the model one catchment is considered as a potential water seller (Tagus River Basin) and the other one as a buyer (Segura River Basin). In no case the roles are reversed (RD 1/2001).

The inter-basin market is activated when the drought index of the “buyer” catchment falls below the emergency threshold and if the “seller” catchment is not in an emergency. In this case, the agents may trade for water and modify their initial irrigation plan to a certain extent in order to increase their revenue. Accordingly, the new objective function of each UDA is defined as follows:

---

\(^3\)Traditionally rainfed crops receiving supplementary irrigation may still produce a yield without being irrigated, and this has to be accounted for (e.g., olive groves, vineyard).

\(^4\)This stochastic variable is obtained from the yield and price historic series (MARM, 2012) as the standard deviation of the revenue, excluding the years with drought.
\[
\text{Max } II = \sum_{i=1}^{N} \left[ \left( P_i + r_i - \sum_{j=1}^{T} c_{ij} \right) * s_i * \Omega(w_i) + u_i + \left( P_i + r_{i,RF} - \sum_{j=1}^{T} c_{ij,RF} \right) * s_i * (1 - \Omega(w_i)) + u_{i,RF} \right] + P * (w_{i_e} - w)
\]

Subject to:

\[
\sum_{i=1}^{N} w_i \leq w
\]

\[
w \geq w_{\text{min}} * S_{\text{ligneous}}
\]

\[-m \leq w_{i_e}\]

\[|m| \leq M_{\text{UDA}}\]

\[
\sum_{k} m \leq W_{ATS} - T_{ATS}
\]

Where \( P \) is the market price of water and \( w = w_{i_e} + m \) is the total amount of water available, being \( m \) the water purchased \((m > 0, \text{SRB})\) or sold \((m < 0, \text{TRB})\) in the water market and \( w_{i_e} \) the water allotment in the considered scenario \( \text{(dependent on the drought index of the previous section, } l_e) \).  

Water resources available need to be at least equal to the amount of water resources required by ligneous crops \( (w \geq w_{\text{min}} * S_{\text{ligneous}}) \). The amount of water available \( (w = w_{i_e} + m) \) has to be at least equal to the total amount of water used by the UDA in the different crops \( (\sum_{i=1}^{N} w_i) \). Also, water markets are limited by water allocation, as no UDA can sell an amount of water greater than its allocation \( (-w_m \leq w_{i_e}) \). In addition, water markets are limited by the capacity of the primary and secondary water canals to transport and distribute water \( (M_{\text{UDA}}) \). Finally, water trade is also limited by the capacity of the Tagus-SEGURA Water Transfer \( (W_{ATS}) \) minus the water transfers outside of the market \( (T_{ATS}) \). The maximum capacity of the water transfer equals 1 000 hm³ per year, though it is limited by law to 600 hm³ per year. During a pre-alert event in the TRB, this amount is reduced to 456 hm³ per year, to 276 hm³ per year during an alert and to 0 hm³ per year during an emergency. The water transfers outside the market are variable, though we use historical data to determine an average value \( (T_{ATS}) \) for every drought event \( \text{(SRBA, 2013)} \).

### 3.3. The water market

The potential buyers in our model are the UDAs in the SRB with a positive surface of non-irrigated crops. Therefore, the willingness to pay for water depends on the water productivity of these non-irrigated crops. On the other hand, the potential sellers are those UDAs in the TRB with a positive water allocation. The willingness to accept for
water depends on the water productivity of the irrigated crops. Accordingly, first of all we need to obtain the productivity of water for every crop \((i)\) and UDA \((k)\) in the irrigated areas \((i/Irr)\) of the TRB \((C_{ik})\) and in the non-irrigated areas \((i/RF)\) of the SRB \((I_{ik})\):

\[
I_{ik} = \left[ (P_{i/Irr,k} \ast r_{i/Irr,k}) - \sum_{j=1}^{T} c_{i/Irr,k,j} - (P_{ik} \ast r_{ik,RF}) - \sum_{j=1}^{T} c_{ijk,RF} \right] / (W_{ik}) \\
C_{ik} = \left[ (P_{i/RF,k} \ast r_{i/RF,k}) - \sum_{j=1}^{T} c_{i/RF,k,j} - (P_{ik} \ast r_{ik,RF}) - \sum_{j=1}^{T} c_{ijk,RF} \right] / (W_{ik})
\]

where \(W_{ik}\) represents the agronomic water requirements in m³/ha of crop \(i\) in the UDA \(k\).

Agents in the model will trade until the marginal cost of water equals the marginal productivity of water. In our unilateral market, the marginal productivity of water equals that of the SRB, while the marginal cost equals the marginal productivity of water in the TRB plus other variables including asymmetric information and transportation and environmental costs. In the next sections we assess all these costs and we obtain the theoretical solution to our model.

### 3.3.1 Basic model with no additional costs

Without asymmetric information and transportation and environmental costs, the marginal cost of water would match the marginal productivity of water in the TRB. In this case, agents would trade until the marginal productivity of water in the SRB matches that of the TRB, i.e., until:

\[
P_1 = I_{ik} = C_{ik}
\]

where \(P_1\) is the market price that equals the marginal productivity of water in both basins.

UDAs will sell or buy water up to the point where the marginal productivity of water is lower or higher than the market price. Therefore, the amount of water traded \((m_1)\) would be:

\[
m_1 = s_i \ast \left( 1 - \Omega(w_i) \right), \ s.t. \ I_{ik} \geq P_1
\]

Or, alternatively:

\[
m_1 = s_i \ast \Omega(w_i), \ s.t. \ C_{ik} \leq P_1
\]
3.3.2. Asymmetric information

In the water exchange there may be some restrictions to access to information. These restrictions may have significant impacts over the marginal productivity/cost as perceived in the market. As a result, the observed price in the water market may not match the “optimum” price obtained in the previous section. These costs are endogenous of each water exchange, and therefore can be represented by a stochastic variable (a white noise $e$ with a standard deviation based on the water prices of previous water markets) (Rey et al, 2011; Calatrava and Gómez-Ramos, 2009). In this case, the water price would be:

$$P_2 = P_1 + e$$

And the amount of water traded in the market would be reduced as compared to the previous section:

$$m_2 = \min \left( s_i \left( 1 - \Omega(w_i) \right) , \ s.t. \ i_{ik} \geq P_2; \ s_i \Omega(w_i) , \ s.t. \ C_{ik} \leq P_2 \right)$$

However this asymmetric information has not yet been implemented in the current model ($e = 0$).

3.3.3. Environmental costs

Water markets may have an impact over the environment, especially in the donor area. In order to prevent environmental deterioration, water authorities may decide that a percentage of the amount of water traded ($env$) must remain in the TRB in the form of environmental flows. To achieve an environmentally neutral water market, these restrictions should be at least enough to compensate for the return flows generated by agricultural water use in the donor basin. Environmental costs increase the marginal cost of water and the water price in the market, and therefore they reduce the amount of water bought. In addition, they reduce the quantity of water received in the SRB, since part of the water bought will be used to satisfy the environmental flows.

$$P_3 = (P_1 + e) / (1 - env)$$

Of this price, farmers in the TRB will perceive only a fraction, since the remaining cost corresponds to the environment:

$$P_{3,TRB} = (P_1 + e)$$

The amount of water traded will be:

$$m_3 = s_i \left( 1 - \Omega(w_i) \right), \ s.t. \ i_{ik} \geq P_3$$
Or alternatively:

\[ m_3 = s_i \cdot \Omega(w_i), \quad s.t. \ C_{ik} \leq P_3 \]

However, the amount of water received by the farmers of the SRB, and therefore used for agricultural production in the SRB will be:

\[ m_{3,SRB} = m_3 \cdot (1 - env) \]

3.3.4. Transportation costs

The Tagus-Segura Water Transfer covers a distance of 242 km between the Bolarque Dam in the TRB and the Talave Dam in the SRB. Therefore, there are significant transportation costs in the form of transportation fees \( f_{TS} \) and losses \( l_{TS} \). This increases the marginal cost of water and the market price:

\[ P_4 = \frac{P_1 + e}{1 - env} + f_{TS} \]

And reduces the amount of water traded in the market:

\[ m_4 = s_i \cdot (1 - \Omega(w_i)), \quad s.t. \ l_{ik} \geq P_4 \]

Or alternatively:

\[ m_4 = s_i \cdot \Omega(w_i), \quad s.t. \ C_{ik} \leq P_4 \]

Transportation costs further reduce the amount of water that reaches the SRB \( (m_{4,SRB}) \) as compared to the total amount of water traded \( (m_4) \).

\[ m_{4,SRB} = m_4 \cdot (1 - env) \cdot (1 - l_{TS}) \]

4. Market scenarios

The inter-basin market is activated when the drought index of the “buyer” catchment falls below the emergency threshold and if the “seller” catchment is not in an emergency (RD 15/2005). So far water markets have worked through bilateral agreements among agents (lease contracts), though in our model we assess the potential outcome of two more scenarios.

4.1. Lease contract

Lease contracts have been the most common legal figure in inter-basin trade. In a lease contract, the potential buyer UDA contacts a potential seller UDA on a bilateral basis. Although water authorities can intervene in order to fix a price or to forbid the water transfer (based on third party effects such as impoverished qualitative status of the aquatic ecosystems or environmental flows reduction), in reality this is unlikely.
and actually public authorities facilitate this type of contracts, for example by offering public subsidies (such as the forfeiture of the transportation fees) (RD 1/2001).

In this scenario, bargaining is bilateral. Therefore, the market is small (only two UDAs in each exchange, \( k = k_1 \) in the SRB and \( k = k_2 \) in the TRB). Transportation costs are included (\( l_{ATS} = 0.1; \ f_{ATS} = 10 \text{ Eurocents/m}^3 \) (SRBA, 2013), but environmental costs are not (\( env = 0 \)) (Rey et al., 2011; RD 1/2001).

### 4.2 Water banks

Water banks are public exchange centres from which no agent can be excluded. In water banks, the river basin authority organizes the market and sets a fixed price (RD 1/2001). In addition, water banks need to take into account transportation costs (\( l_{ATS} = 0.1; \ f_{ATS} = 10 \text{ Eurocents/m}^3 \)) and third party effects to prevent environmental deterioration. This means that water banks must impose restrictions on the amount of water that can be traded from the TRB to the SRB. These restrictions should be at least enough to compensate for the return flows that would be otherwise lost in a market. Return flows are estimated at 19% in the UDAs of the TRB with access to the water transfer (\( env = 0.19 \)) (SRBA, 2013).

### 4.3 Water banks and lease contracts

In this scenario an institution represents all the irrigation communities from the SRB (buyer). This institution contacts a potential seller (the UDA, \( k = k_1 \)) and they negotiate a price and a volume of water to be exchanged. There are also environmental restrictions and transportation costs.
a- Water bank scenario

b- Lease contract scenario

c- Water bank lease contract scenario

Figure 4: Opportunity for water trading in 3 different scenarios: water bank, lease contract, water bank lease contract
4.4. Outputs of the model

For each scenario if relevant the model assesses:

- The revenues in Euros per each UDA and in total
- The volume of water allocated to each UDA
- The surface of crops irrigated and non-irrigated.
- The price per contract

As each event is independent and results of a stochastic approach, expected revenues can also be calculated for each scenario and compared.

5. Results

Hundred simulation steps were run with the model to obtain the following results, one simulation step representing a single year with a different Tagus and Segura drought levels obtained by the stochastic approach each time.

Table 2 shows the average revenue obtained in each catchment for the different levels of drought. The revenues are for the baseline scenario, i.e. without any trading market in place. Expected losses for the Segura catchment are higher than in the Tagus catchment. Thus, during an emergency event, loss in revenues due to irrigated water volume restriction are up to 42% of the average normal revenue in the Segura catchment, but only 20% in the Tagus catchment. The differences between both catchments may be explained by the difference in land uses and by the variability in irrigated-rainfed net margins factors.

Table 2: Changes in revenues (€) under baseline scenarios

<table>
<thead>
<tr>
<th>Drought</th>
<th>Segura Revenues €</th>
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<tr>
<td>Normal</td>
<td>490 643 602</td>
<td>Normal</td>
<td>44 410 163</td>
</tr>
<tr>
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<td>473 516 114</td>
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In Figure 5 the expected changes in revenue for the different scenarios (baseline and trading markets) under an emergency event in the SRB are compared. Overall, any type of contract increases total welfare (measured by total income) as compared to the baseline scenario. Yet the reader needs to keep in mind that the consequences in terms of local employment are not considered in the model. The “water bank” option is the most effective one, yet very similar to the “water bank lease contract” option. In the “water bank” scenario the Segura catchment is better off at the disadvantage of the Tagus catchment. For these two scenarios it is important to consider that a hundred per cent participation is assumed in the model. Therefore lower revenues may be expected. For the lease contract 20 contacts were used as a basis. Further research is necessary to explore the participation process and adjust the results of the model on this basis.

The drought level in the Tagus impacts on the increase of revenues in the Segura catchment as less water is available for trading. In addition, the lack of water means that only highly productive water is available for trading, thus increasing the selling price. Interestingly, the Tagus catchment benefits slightly of the emergency situation in the water bank scenario. This can be explained by the rise of the price value in an emergency situation. Indeed for a “water bank” scenario price is set up at 0.49€/m³ in a normal situation, at 0.5€/m³ in a pre-alert situation and at 0.64€/m³ in an alert situation (including transportation fee).

In the “lease contract water bank” scenario the variability of the price value is high with a majority of values around 0.90 €/m³ (Figure 6). As illustrated on Figure 4c the main reason is in the difference of demand and supply, as all Segura UDAs are grouped and bargain with each Tagus UDA separately. In the lease contract approach prices varies from less than 0.15 €/m³ to 0.95 €/m³ with a predominance of low price values.
In both types of contract however asymmetric information and an adaptive strategy during the negotiation process may influence the final price. For instance the fact that all the Segura UDAs are grouped under one institution may allow them to better negotiate and to reduce the price. As stressed in 3.3.2 section adjusting a white noise variable (e) may improve the model results.

![Figure 6: Water Bank Lease Contract Prices (fee included)](image)

![Figure 7: Lease Contract Prices (fee included)](image)

In Figure 8 and 9 the variation in the increase of revenues between UDAs are represented. The opportunity for the UDAs to benefit of the market depends of the surface of crops with high water productivity for the Segura and low water productivity for the Tagus. In the Segura some UDAs such as UDA number 6 cannot
participate in any transaction due to its low productivity crops. An UDA is composed of different irrigation communities, modelling at the level of the UDAs may therefore hide further disparity between communities.

Figure 8: Increase of revenue compared to the baseline per UDA in the Segura

Figure 9: Increase of revenue compared to the baseline per UDA in the Tagus
6. Conclusions

The EPI4Drought agent based model has been developed by FHRC (MU) and IMDEA to assess and to compare the potential of water markets to attain a better allocation in the particular case of the Tagus and Segura interconnected river basins in Central and South-Eastern Spain. The allocation of water to the different crops and the market price are mainly ruled using the water productivity concept, defined in this project as the irrigated productivity minus the rainfed productivity divided by the water requirement for different groups of crops. The UDAs (agricultural demand units) entities with available information on both catchments have been used as agents in the model. The model simulates independently annual events, the main stimuli being a change in the drought level defined stochastically on each catchment. The model compares three different types of market to a baseline scenario: water lease contract, water bank scenario and a water bank lease contract. Overall the three options increase the welfare in both catchments as compared to the baseline. The water bank lease contract scenarios and the water bank scenario provide similar total revenues. Defining which of the two scenarios is the preferred option is more elusive as distributional effects differs at inter and intra catchment level. The water prices are dependant of the type of market. Not all the UDAs are benefiting of the market situation.

The model is still at an early stage and further research will aim at improving it including:

- Collecting data at a lower level such as the irrigation communities
- Modelling the potential strategies in the bargaining process
- Indirect impacts of the market on the local economy
- Shorter duration in the model such as monthly steps to better represent the crops water requirement and adaptive behaviour
7. References


Royal Decree (RD) 1/2001, de 20 de julio, por el que se aprueba el texto refundido de la Ley de Aguas.

Royal Decree (RD) 15/2005, de 16 de diciembre, de medidas urgentes para la regulación de las transacciones de derechos al aprovechamiento de agua.


Water Act 29/1985, de 2 de agosto, de Aguas.

Water Act 62/2003, de 30 de diciembre, de medidas fiscales, administrativas y del orden social.
Research Task 4.2 Output 14

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Drought Management in England

16th July 2013

Grant Agreement no. 265212
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Colin Green - Flood Hazard Research Centre – Middlesex University

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Introduction

We manage water in order to make the best use of land and the usefulness of land is heavily influenced by the reliability of access to water. In general, the higher the value of the activity on the land, the more reliable must be water availability to that plot of land. In arid climates, the value of land is largely given by the reliability of access to water.

The problem in water management is to manage the inherent variabilities in water availability and in demand. The greater the inherent variability in availability, the greater the difficulties of achieving a match between availability and desire. Hence, from the perspective of Integrated Water Resource Management (GWP 2000) droughts and floods are no more than the two extremes of this variability and not free-standing problems (Technical Support Unit 2003). In arid climates, those characterised by extreme variability, floods are the water resource. The problem is then to determine how much variability should be considered since the resources invested to cope with those extremes will be non-productive the rest of the time.

![Annual rainfall averages](image)

*Figure 1 Rainfall patterns (data source: Henson 2002)*

Variability is characterised by both intra-year component, the seasonal pattern, and the inter-year component. Countries differ very markedly in the seasonal pattern of
rainfall (Figure 1). In most countries, there is either a very marked ‘rainy’ season or there is little variation in rainfall over the year. A few countries are characterised by two wet seasons and a dry season. Since the biggest demand for water is by plants, which largely use water for cooling, a further difference between countries is the timing of rainfall relative to the growing season. That transpiration demand by plants (Potential Evapo-Transpiration) is determined by microclimatic factors, including air temperature and wind speed, as well the nature of the crop. Whilst soil has a capacity to hold water in the vadose zone for use by the plants, this capacity depends upon the nature of the soil and hence the number of days without rainfall or irrigation that the plants can survive varies from place to place. Marked gaps between PET/P have to be bridged by irrigation if plants are to survive.

Inter-year variability in rainfall differs significantly between countries (Figure 2). An understood cause are the oscillations, notably the ENSO event in the Pacific ocean, but there is also a North Atlantic Oscillation.

![Variability of annual rainfall chart](chart)

*Figure 2 Inter-year variability in rainfall (data source: FAO Aquastat statistics)*

That variability is inherent means both that there is no consistent formal definition of a drought between countries and that what constitutes a drought varies from place to place. In different parts of the world, the likelihood of a reduction of, say, 40% in the availability of water for a given period of time shows marked differences. In particular, the duration of a below average period of water availability varies markedly. Thus, droughts of 5-7 years duration have been experienced in California and Australia. In turn, Sydney has 3 years water demand in storage. Conversely, London has 90 days supply held in reservoirs (Green 2003) since a 5 year duration drought is considered unlikely.
In addition, for practical purposes, a drought has to be designated in prospective terms; as a signal to take action to respond by making better use of water resources to an anticipated mismatch between demand and supply. The longer a current shortfall might be anticipated to last, the greater the requirement for managing demand in the short term. The existence of patterns such as ENSO make drought forecasting more practical.

**Context**

Water usage is always dominated by plant requirements and hence whether the ratio of Potential Evapotranspiration to precipitation (PET/P) means that arable irrigation is possible within a particular area without irrigation. In England, although rainfall is low in the south-east of the country, so are PET requirements; consequently irrigation has never been essential. Similarly, the practice of rainwater harvesting was also traditionally very limited in contrast to Spain. As far as is known, the only tradition of rainwater harvesting in England was the use of ‘dew ponds’ in parts of the country. The name is a misnomer; these were simply clay lined ponds which collected runoff from the hills above. The predominant use was then for livestock watering.

England has a temperate climate dominated by the Atlantic. Inter- and intra-year variability of rainfall is low although average rainfall in parts of the country is low (in the south-east and London it averages 600 mm). In parts of the country, a large proportion of runoff (‘blue’ water) is already put into use (again true of the south-east). Whilst in some regions, there is a groundwater resource to be tapped, the predominant source of water is surface water either from impoundment reservoirs or by the direct abstraction from rivers. Rivers are small in terms of average flows and flows are unsupported (i.e. not dependent upon discharges from reservoirs). That rivers are unsupported is a marked difference from some other countries notably Australia, the western USA and Spain where markets have been created essentially in the shares of storage in reservoirs. Over abstraction of groundwater from chalk aquifers has created a number of severe low flow problems in rivers whose primary flow is from aquifers.
When the water and wastewater industries were privatised, the existing capital assets were simply transferred to the companies that were to be privatised, the existing organisation of the industries was not changed except that these organisations were now shareholder owned companies. The new companies maintained, therefore, the largely catchment basis of the previous organisations; the distinction is that whereas the wastewater organisations were wholly catchment based, the rather more fragmented state of water supply was not amended.

In England, agriculture is a mixture of livestock and arable farming; grass being the crop with the largest area (Figure 3). Because it has a temperate climate, agriculture is almost exclusively rainfed with only supplementary irrigation being used and for high valued crops (supplementary irrigation is intended to match the delivery dates and quality requirements of the supermarket chains). One consequence is that irrigation is almost exclusively spray irrigation, since the equipment can be moved from field to field as required. High efficiency irrigation, notably drip irrigation, is rare and flood irrigation was not practised (the exception being the creation in some parts of the country in the seventeenth century of water meadows: these were lowland areas where there was engineered surface water flooding in winter, one of the purposes being to warm the soil and hence promote early growth of grass).

Hence, the dominant uses of water abstraction are for urban uses (Figure 4). However, many catchments are already over-abstracted and there is over-abstraction from aquifers which are available in some parts of the country. Figure 5 gives the number of licences for different uses in the different regions. What this shows is that although there are a large number of licences for agricultural usage, the average abstraction is tiny compared to that for any other use.
Hence, potential exchanges of water in England are largely limited to between fossil energy supply and public water supply or within each sector. Both uses are essentially non-consumptive, the majority of water used in each case being returned to the waterine environment, in marked distinction to irrigation. Indeed, urban areas typically export more water than they import as urban areas are highly effective systems of rainwater harvesting by reason of their impermeable areas (Green 2003).

The potential of beneficial exchange, which might be exploited by the use of EPIs, is determined by extent of the differences that exist. That the peak demand for energy is normally in winter whilst that for water is in summer creates some potential for exchange between the public water supply and energy sectors. The shift to gas fired generation and more especially out of fossil fuels will create a much large potential for the shifting of water from steam production and cooling (Meldrum et al 2013) to public water supply but, in both cases, only of that fraction of the water used in energy production which is consumed in the process.

The potential of exchange within each of the two fields is unclear but the energy costs of transfer between catchments means that the potential is likely to be greatest in the lowland parts of England and where privatisation did not result in catchment based water companies.
The 2012 drought in England

A ‘drought’ can be defined in one of two ways:

1. For rainfed agriculture, when short term PET exceeds local precipitation.
2. For urban areas, when long term precipitation falls over a substantial period below averages so that runoff – and also infiltration to the aquifers – falls well below the average.

A drought was declared in April 2012 on the second basis; precipitation having been below average for two years. The declaration of a drought was almost immediately followed by heavy rains. In turn, agricultural production was damaged not by the scarcity of water but by the excess water. The 2012 drought was the first major and widespread drought since that in 1976; the event in 1976 occurring prior to water privatisation. A localised drought in Yorkshire in 1995 had exposed that privatisation had removed the political cover for the water industry. In 1976, the then government appointed a Minister to take charge of co-ordinating response to the drought. Conversely, in the Yorkshire drought, the Minister came back from his summer holiday to hold a meeting with the Chief Executives of the water companies in order to issue a press release saying that their performance was unacceptable. In turn, although Yorkshire had been installing stand pipes, the company had to truck in water by lorry from outside of the drought area.

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**Figure 5** Number of abstraction licences in different regions (Source: https://www.gov.uk/government/statistical-data-sets/env15-water-abstraction-tables)
A key element in the response to the 2012 drought was the knowledge that the London Olympics and ParaOlympics would be held in the summer. It would have been humiliating if there were severe water use restrictions in place during the Games. This nightmare scenario had previously been foreseen and was probably the reason why a desalination plant had been built at Beckton to support supply in east London. This plant is purely a drought contingency measure and has never been run. It was financially viable for Thames Water to build the plant because of the structure of price regulation for the industry. Thames Water are allowed to immediately charge the designated appropriate return on capital for the capital cost of the plant to consumers' bills irrespective of whether the plant is being used. However, the regulatory structure also requires 'efficiency' improvements in O & M costs and since the O & M costs of water from the plant are higher than from alternative sources, it will be operated only in extreme conditions.

What is of interest therefore in 2012 drought is the responses that were proposed to the immediate threat and the proposals for future drought risk management that followed from the perceived 'crisis'. What is a ‘crisis’ is always difficult to define; as a working definition it might be termed an event which results in a search for alternatives to the present courses of action. A crisis is thus an interpretation rather than an event. For any change to then occur, there then has to be at least one apparently feasible alternative to the cause of action then being adopted. Two major alternatives for future action that were then discussed during the 2012 drought were:

- inter-basin transfers of water
- demand management

I should declare an interest at this point in that I was all over the media arguing that we had to use water much more efficiently than we had in the past and that the 30% reduction in demand would bring us down to best practice levels in other countries would create sufficient headroom for droughts.

The main alternative strategy proposed to demand management were inter-basin transfers. Since the water companies are largely organised on a catchment basis and as a monopoly within a large catchment, this necessarily involves inter-company transfers. Some inter-basin transfers were constructed prior to privatisation and sub-catchment transfers within individual company areas have been constructed post-privatisation (Harou 2010). The idea behind the proposal is the differences in both water availability and demand in different parts of the country. This idea has been considered in the past (Detr 2000) and more recently (Cave 2009; Ernst and Young 2011; Frontier Economics 2011; OFWAT 2010; Raffenperger and Harou 2012). The two major drawbacks are the costs of doing so and the ecological consequences if water is transferred from water course or water body to another. The inherently cynical economist anticipates that wherever there are long distance transfers, particularly where the transfer is not solely by gravity, then there is a massive subsidy involved. The existing price regime for the water industry does encourage
such transfers within a company’s area because once the investment has been agreed by the price regulator, the company is guaranteed a fair return on the capital invested.

Inter-company transfers are more problematic in that this would require either creating new licences for abstraction, that is assuming that the water is available, or the transfer of existing abstraction licences from one company to another. The Cave Report, prepared for the last government, proposed the introduction of a system of tradeable water abstraction licences and also retail competition in water supply. But the Cave Report had two major drawbacks:

1. a failure to understand the economics of water management, and
2. an assumption that demand will increase in the future.

Water and water management has a multiplicity of characteristics which make it very different from the resource assumed in micro-economic textbooks. Amongst these differences are that capital intensity results in physical economies of scale and short run marginal costs seldom rise with demand but may be constant or fall. This makes it difficult to introduce competition. That water is heavy and incompressible means that strategies that work in the telecommunications or electricity industries are unlikely to be effective in the water industry.

An example of this failure of understanding is the example often quoted both during the preparation of the Cave report and subsequently. This was that multi-site companies, such as supermarkets, wanted to be able to contract with a single supplier for water and wastewater services in the same way that they do now for other utilities. But realistically it is never going to be financially viable for a site in the south-east of England to buy in water from the north-west of the country and discharge its wastewater to the south-west of the country for treatment. Focusing on trading and competition directs attention away from some of the readily treatable problems. The approaches advocated in the Cave report also offered minor efficiency gains. Conversely, for example, one of the problems in practice is that the privatisation legislation left water companies billing addresses and not owners of properties. Hence, a company with 1500 properties will receive 1500 water bills, one for each address. More generally, some of the water and wastewater companies behave like archetypal monopolies lacking any sense of customer service.

Secondly, the report assumes that demand will increase whereas the policy of the current and previous government has been that sustainable water management shall be introduced with the immediate aim of reducing water consumption in 2030 to the levels in Germany now. The problem therefore is how to induce this change; in particular, how to provide incentives for the companies to promote sustainable water usage. Part of this requirement is to demonstrate that the companies will be able to
make profits when consumption has fallen as well as from promoting the change itself.

Whilst demand per person is intended to fall, domestic and industrial uses being 30% or so above best practice levels (Green and Anton 2012), unlike Germany and most of the rest of Europe, the current population projections for England are for a growth in population (Giannakouris 2008). The problem with all population projections is that they are usually wrong; in the 1920s, the Metropolitan Water Board was projecting that in the 1960s it would have to provide water to a population in London of 20 million people (Chevalier 1953). Similarly, in 1960s and 1970s a number of reservoirs were developed, notably Kielder Water (the largest reservoir in the UK), in anticipation of demand which did not materialise (McCulloch 2006).

In considering trading, a starting problem is that the Water Resources Act 1961 grandfathered all existing licences in perpetuity. Whether this was done intentionally or simply because the legislation was not thought through and the desire was simply to avoid re-licensing all the existing licences is not clear. All subsequent licences are time limited. There is as a result a large overhang of sleeper licences: unused licences for which there is an entitlement to use (Figure 6). This is in addition to the over-allocation of some rivers and aquifers. A concern to be taken into account in considering trading is the risk that sleeper licences would simply be activated, a risk especially when a catchment or aquifer is already over-abstracted.

Figure 6  SLEEPER LICENCES FOR WATER ABSTRACTION IN ENGLAND AND WALES (BY AMOUNTS)


There are two obvious reasons for holding an abstraction licence but not using it. The first is that access to water is a necessary pre-condition for investment in a thermal electricity generating plant. Without access to water for steam generation
and cooling (although in South Africa, 'dry cooling' techniques have been developed to reduce the cooling requirement), the investment is dead. Hence, an appropriate abstraction licence is necessary before any company would contemplate constructing a new thermal plant; that licence probably needs to provide security of supply for 25-30 years at a minimum. There is no readily available data on these sleeper licences so it is not known whether these licences are left over from thermal plants which have been decommissioned, issued for plants which were never constructed or for expansions which were never undertaken.

The second logical reason for holding licences for more water abstraction than is required is to provide security of supply: to hold licences for abstraction from different sources so that if a drought or pollution limits the availability of one source, abstraction can be undertaken from another.

There are three forms of water resource:

1. supported rivers i.e. those whose flow rate depends upon the release from an upstream reservoir
2. unsupported rivers
3. aquifers

Trading might take place either within each of these resources or between them (for example, as part of conjunctive use of surface water and an aquifer). In the first case, trading is effectively in shares in storage and in the latter two cases of shares in runoff and recharge respectively. Again, in the first case, what is available depends upon the storage capacity and in the last case upon the sustainable yield. In unsupported rivers, in most cases, available abstraction responds very quickly to changes in runoff and hence in rainfall. Some, generally small rivers, have their base flow supported by outflows from aquifers where the time lag between precipitation and discharge may be of some length. This lag has promoted the adoption of conjunctive use schemes where suitable aquifers exist: aquifers are recharged from surface water, the aquifer being used as a storage reservoir. One example is the North London scheme where the aquifer is recharged in winter by flows brought down by the New River. In summer, water is pumped from the aquifer and the New River is used to transport that water, by gravity, to treatment works.

For rivers, trading has a directionality: up- or down stream. For an aquifer, trading depends upon the depression cones created. Trading might be within a catchment or more problematically between catchments; the latter potentially creating ecological problems (e.g. differences in the pH of the source and receiving waters) as well as the cost problems and there is an additional risk of spreading invasive species. In England, because the water supply companies are largely organised on a catchment basis, trading would largely have to involve inter-basin transfers so the transfer costs will tend to be high.

**Figure 7** is a hypothetical example of a supported catchment where water is used for irrigation in four blocks. It is a closed catchment; all available water is fully utilised.
so that the river runs dry in some areas and less than 1% of the water reaches the sea. In this illustration, improving the efficiency of water usage (e.g. switching from spray irrigation to drip irrigation) or re-allocating water between the irrigation areas produces minimal gains of less than 2%. For more significant gains therefore, it is necessary that there be differences in the returns to water between the areas; this is where the gains from trade have occurred in Australia. But it should be noted that in reality the value of the water in the river will also fall as it travels downstream since it consists largely or entirely of irrigation drainage water and thus contains increasing proportions of salts.

Figure 7 Abstractions in a hypothetical supported catchment

What this example also illustrates is that the instream value of water will influence what is the most efficient allocation of water between areas. Using arbitrary values for instream water, but ones which environmental groups and NGOs might argue
are relatively rather low, the option which involves the largest gain is that which simply reduces upstream abstraction and reallocates that abstraction downstream. But the gain is almost entirely from environmental benefits. In order to produce any incentive for the upstream abstractor to trade with the downstream user, it would be necessary to introduce environmental charges on abstraction. That looks to be a complex problem. Otherwise gains may arise because of differences in the return to water from different crops; this is notably the gain in Australia where trades have been from crops with a low return to water to those with a high return to water (Frontier Economics/Tim Cummins and Associates/Institute for Rural Futures; Zuo et al 2012). But there is a directionality issue; from upstream to downstream or vice versa and also that crops, like all plants, fill ecological niches; an area suitable for one plant type not necessarily being suitable for another.

For unsupported rivers, the problem is more difficult. Here, river flows vary seasonally but out of step with the seasonal variations in demand. In England, water demand is higher in the summer than at other times in the year whilst in consequence of the increased PET, runoff and hence flows are lower in the summer than in the winter.

![Graph showing hypothetical unsupported catchment with two cities](image)

**Figure 8 Hypothetical unsupported catchment with two cities**

In Figure 8, cities A and B lie on an unsupported river; the two curves show the pattern of demand for each of the two urban areas. It can be seen that it is likely to matter which is the upstream city (shown in the figure as A) and what is the return fraction of water. In addition, since urban areas are effective systems of rainwater harvesting so the surface water drainage produced in each urban area needs also to be considered. The obvious answer is to introduce storage either by partially supporting the river or in the form of offline storage at the point of abstraction (e.g. the case in London). The normal economies of scale mean that, if the physical
conditions are able to support storage, then building one reservoir to support both urban areas is likely to result in lower per unit costs than two reservoirs. In addition, there is the problem of providing space for the required storage; lowland provision being likely to require a greater area than an upland site. Hence, the traditional practice of cooperative action between municipalities; the verband in Germany and syndicat in France for example.

At the start of the 2012 drought, Severn-Trent Water offered to sell 30 million litres of water to the neighbouring Anglian Water company area, subject to negotiations about price (Water 21 2012). Whether this was a real offer or a political gesture is not clear. What would have been an appropriate price is also unclear. The water would have been pumped from a group of boreholes developed by Severn-Trent Water as a contingency measure against a drought (an example of a sleeper licence). Under the price regime for the water industry in England and Wales, Severn-Trent Water would first have had to get the development of those boreholes and associated infrastructure approved by the price and service regulator (OFWAT). Once the development was approved, Severn-Trent became entitled to earn the set appropriate return on the capital invested (currently 6.5%) and to increase charges to their consumers appropriately. Thus, since the boreholes were developed, Severn-Trent customers have been paying for them. It might be argued therefore that the customers should be entitled some or all of any sales revenue from Anglian Water in excess of the O & M costs of producing and transferring the water. But if customers were to take all of this return then there would be no incentive in this or similar cases for the water company to sell the water to another company. That the drought was rapidly brought to an end by heavy and continuous rain meant that the proposal was abandoned.

This example may be a special case. In general, for one water company to offer to transfer water to another, the logic is that this should be a purely commercial decision for both parties. Thus, the potential supplier should not be part of regulated water company and hence not be guaranteed a return on capital from its own consumers but be free to make a commercial decision as to whether to make the necessary investment, generating the necessary return by exports. Secondly, the capital costs of developing the resource and of the inter-basin transfer are likely to require that the resource is used permanently in order to generate an adequate return on the capital employed. Thus, the potential purchaser is likely only to be prepared to purchase the water when it can be used as essentially a baseload source of supply, displacing another source whose O & M costs are higher. That saving then can be argued to belong in part to the consumer of the importing company who is currently paying the O & M costs of supply from the existing source and, if that resource has been developed post-privatisation, a return to the capital employed in developing that resource. But if all those savings were returned to the consumer, there is no apparent saving to the potential importing company of importing the water. More generally, there is an on-going debate as to how to restructure the price regulatory
system for the water and wastewater industry to promote both efficiency and sustainable water management (Severn-Trent Water/National Grid 2012).

**Drought management in England**

Given that the requirement in water management is to manage variability, the long term requirement is to increase the capacity to do so. Where this capacity proves to be inadequate, the remaining two options are:

1. to increase the intensity with which remaining resources are used;
2. reduce demand to meet sustainable supply

In extremis, the means of increasing the availability of water is the desalinisation of sea water. This comes at a high energy penalty both in desalinisation itself but also in raising the water to the heights of the demand centres. The development of graphene as the membrane would reduce the energy requirements of desalinisation itself but not reduce the costs of moving the product water. Hence, the primary role of membrane technology may lie in the treatment of blackwater near the point of use, allowing its recycling, rather than in the production of potable water from salt water. Recycling and reuse are two means whereby the intensity of urban use of water resource can be increased; for example, the increasing use of local greywater recycling plants in Japan (Gaulke 2006) and the use of a membrane treatment plant to take blackwater directly from a trunk sewer and deliver it, after treatment, for such uses as irrigation of parkland (ODA 2012).

In European terms, England does not use its water resources efficiently, a reduction in around 30% being required to match current household consumption levels in Belgium (105 l/p/d) and 20% to match levels in Germany (Green and Anton 2010).

To meet extreme conditions, each water company is required to prepare a Drought Management Plan which must be approved by the Environment Agency (Environment Agency 2008, 2012). Unlike the requirements in California for such plans (Colorado Water Conservation Board 2010), there is no specific requirement for the company to consider the implications of a drought for revenues and costs. In this regard, England is lucky in that domestic water metering is relatively modest in penetration so that the revenue risk from either droughts or long term reductions in demand is limited. Conversely, where metering is used, a drought is immediately reflected in a fall in revenues whilst costs fall to a lesser degree.

In the extreme, the six mechanisms by which demand can be reduced to meet available supply are rationing by:

A voluntary action: this is surprisingly effective with reductions in demand of 20% plus being achieved (Green 2003).
price: the main use of prices globally in droughts seems to have been the use of punitive tariffs for those households deemed to be continuing to use excessive amounts of water (they are used in this way in South Africa).

effort: in extreme conditions, mains supply in England is cut off and supply is limited to standpipes in the street and water tankers. Demand is thereby limited to what people can carry and so discriminates against the elderly, the disabled and those living in apartments without lifts.

quantity; notably, the use in Zimbabwe and South Africa of flow restrictors on supply pipes (i.e. the fitting of 12.5 mm cuff on to a 15mm pipe) or the reduction in the pressure head of supply (with the consequent problems for high buildings).

time: supplying water for only limited times during the day. The experience in Zimbabwe is that creates both purity problems (infiltration into the water mains and pipes) and creates blockage problems in the mains as well as pressure shock damaging pipes (Green 2003).

prohibition of those uses deemed to be of low priority (Environment Agency 2008): where domestic car washing is permitted then this is one use that is commonly banned. In England, the first level of prohibition is on the use of hose pipes (historically, the use of hose pipes was excluded from those domestic uses which companies had a statutory duty to supply).

Banning external uses provides an immediate and effective means of reducing domestic water consumption. Figure 9 shows that water consumption varies by income (the ACORN social classification is primarily income based), by season, and between weekdays and weekends. Using pricing would require universal metering, the two primary drawbacks of metering being:

the additional cost of metering (in England, adding about 20% on the cost of water supply for simple meters (OFWAT 2011), and more for ‘smart’ meters which could be used for seasonal tariffs). Consequently, it is never economically efficient to meter everyone since low users can never reduce demand sufficiently to cover the additional cost of metering (Green 2003).

metering creates a revenue risk both when demand is driven down in response to a drought and when demand falls over the longer term. In these conditions, costs fall slower than do revenues. Consequently, metering should not be introduced when it is anticipated that demand will fall but only when it is believed that demand will continue to rise.
Drought orders also prohibit some agricultural uses as well irrespective of the nature of the licence that is held (Environment Agency 2008).

**Conclusion**

What this discussion has made clear is that the climatic and demand conditions in England and Spain are very different. Hence, the scope for transposing the approach in England, managing a problem for urban consumption, to the Spanish situation, where demand is driven by the requirements for irrigation, is limited. But there are some general lessons:

- Consider droughts in terms of the overall problem of managing variability.
- Frame drought management from the wider perspective of Integrated Water Resource Management.
- As with floods, the asset created to cope with the extreme is potentially lying idle and generating no return most of the time. In consequence, strategies which produce some return during non-drought periods (e.g. conservation agriculture, rainwater harvesting) are likely to be preferable. Interventions solely against extreme events are unlikely to be economically viable.
- Any crisis presents an opportunity for promoting a shift towards sustainable water management and away from the traditional approach of simply chasing demand ever upwards.
• In urban uses, voluntary reductions in urban demand typically average 20% in the USA (USACE 1995) and UK (40% in South Africa (Millar 2012)), but this is the consequence of squeezing out existing inefficiencies. One consequence is that the transition to sustainability, with its emphasis on using resources, such as water, more efficiently will reduce resilience to shocks such as droughts: there will be less inefficiency to be reduced in response to the shock.

• Longer term, droughts provide an opportunity to drive out inefficiencies in current urban demand. The drought in Barcelona provided an incentive to drive down household demand to 105 l/p/d (Sauri 2009).

• The general finding for water management is that prices are ineffective in reducing demand, a finding that holds for urban uses (Dalhuisen et al 2001; Rees 2009) as well as agricultural uses (Cornish et al 2004; Molle and Berkhoff 2007). This is a fascinating question for economics but the practical problem is to find means of managing demand that do work.

• However, the logic from an economic perspective is to introduce full cost recovery before considering other measures such trading. The trading of subsidies does not increase economic efficiency although it may achieve other objectives. Notably, the highly successful water trading programme in Australia (Young 2010) takes place in a context which does not comply with the requirements of the Water Framework Directive. ‘Full cost recovery’ is not applied, although in most areas, recurrent operating costs are recovered (Parker and Speed 2010). There is no attempt to recover the capital costs of existing storage which costs were borne by the general taxpayer. Nor is it clear whether the capital costs of any new inter-basin transfers would be borne by the irrigators. Similarly, improving the ecological quality of the rivers is being achieved by the purchase of abstraction entitlements, at the anticipated cost of A$3.1 billion for 2750 GL of surface water abstractions, by the Australian Federal government, again instead of applying the principle of full cost recovery (Parker and Speed 2010).

• The advantage of a drought as a problem is that there is a long lead time, the problem being when to anticipate that there will be a problem some time in the future. This implies that a drought forecasting function is essential. Early rather than late designation of a drought allows for the adoption of a rather more tailored response.

• When a crisis has developed, quick fix solutions are sought, notably desalination plants and where desalination plants are constructed, it is common that they are then left mothballed. This is true of a number of plants constructed in response to the last drought in South Africa (Millar 2012) and is apparently also true in Australia and Spain. This suggests that package and hence portable desalination plants are a potential strategy for crisis situations but permanent plants have a limited utility. One problem being
that water then has to be transferred to the points of demand uphill from the coastline so incurring a further energy penalty.

- From an economic perspective, inter-basin transfers usually illustrate Reisner’s remark that water runs uphill to political power; and typically involve substantial subsidies.

- Globally, the biggest losers in a drought are the environment and agriculture.

- The results of an earlier review of the use of tradeable abstraction licences in England (DETR 2000) was there were easier gains to be had from promoting winter abstraction by farmers from rivers and the use of on site storage by farmers. Further gains were to be had by relaxing the definition of the abstraction licences by eliminating some of the definition of the exact purpose of the abstraction e.g. for spray irrigation of a particular field.

- The obvious lessons are the need to set potential abstractions at sustainable levels before introducing any trading and to have an effective monitoring and enforcement strategy in place. That needs to take account of the inherent errors in metering and the tendency of meters to become more inaccurate over time (Arrequi et al 2005). Apparent falls in demand as a result of metering may simply reflect the tendency of meters, as they age, to increasingly under-record flows.

- In arid areas, the value of the land is largely determined by the reliability of access to water. In turn, development of that land will only occur if there is reasonable certainty that water will be available on a reliable basis. The problem of ‘sleeper licences’ then is that long term investment will necessarily require an abstraction entitlement before committing the investment. As shown in Figure 6, a large proportion of sleeper licences in England are for energy production. One implication is that no catchment should be fully allocated; there should always be a reserve for development. A second driver for sleeper licences is likely to be a contingency against the normal source not being available; for example, because of a drought or as the result of short or long term pollution. However, in England, the nature of sleeper licences does not appear to have been fully researched.

- However, it is necessary to recognise the risk that if a trading system is introduced that sleeper licences will be traded and used.

- Where abstractions are supported (i.e. depend upon releases from dams) the obvious logic is to introduce full cost recovery before introducing trading in what is effectively shares of storage.

- A danger otherwise in trading is that it will otherwise create a demand for future subsidies (e.g. when existing reservoirs reach the end of their life). The more apparently permanent the tradeable entitlement, the greater will be the presumption of a future subsidy. Given that agriculture is almost invariably subsidised in some way, this is a real risk.
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