INSTRUMENTOS ECONÓMICOS PARA LA GESTIÓN DE LOS RECURSOS HÍDRICOS EN ESPAÑA

Tesis doctoral

Nazaret Mª Montilla López

Córdoba, febrero 2018

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ECONOMIC INSTRUMENTS FOR WATER RESOURCES MANAGEMENT IN SPAIN

Nazaret Mª Montilla López

Dirigida por:

Dr. D. José Antonio Gómez-Limón Rodríguez
Dr. D. Carlos Gutiérrez Martín

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por compendio de artículos con mención internacional

Nazaret Mª Montilla López

Directores

Fdo.: José A. Gómez-Limón Rodríguez

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Córdoba, febrero 2018

TÍTULO DE LA TESIS: Instrumentos económicos para la gestión de los recursos hídricos en España

DOCTORANDO/A: Nazaret Mª Montilla López

INFORME RAZONADO DEL/DE LOS DIRECTOR/ES DE LA TESIS

(se hará mención a la evolución y desarrollo de la tesis, así como a trabajos y publicaciones derivados de la misma).

La doctoranda inició su colaboración con el grupo de investigación *Water, Environmental and Agricultural Resource Economics* (WEARE, Grupo del Plan Andaluz de Investigación –PAIDI– SEJ-592) en septiembre de 2014 para la realización de su Trabajo Fin de Máster, trabajo con carácter de investigación realizado en el seno del proyecto del Plan Nacional de I+D+i AGRIGOBERSOS ("*Indicadores sintéticos de sostenibilidad: un instrumento para la mejora de la gobernanza del sector agrario andaluz*", AGL2010-17560-C02-01). Posteriormente obtuvo un contrato predoctoral (FPI) en la convocatoria de ayudas para contratos predoctorales para la formación de doctores del Subprograma Estatal de Formación del Ministerio de Economía y Competitividad, asociada al proyecto de investigación del Plan Nacional de I+D+i MERCAGUA ("*Diseño de nuevos mercados de agua para España: Evaluación como medidas para la mejora de la eficiencia en su uso y la adaptación al cambio climático*", AGL2013-48080-C2-1-R). El periodo de disfrute de la beca comenzó en abril de 2015, permitiendo que la doctoranda llevase a cabo su investigación doctoral.

Dicho trabajo de investigación se presenta como compendio de publicaciones dado que ha producido resultados que han sido sometidos a procesos de revisión por pares en revistas científicas de impacto JCR y en un libro de una editorial de prestigio. A continuación, se enumeran las publicaciones derivadas de la tesis:

Artículos científicos en revistas indexadas en el Journal Citation Report (JCR):

- Montilla-López, N.M., Gutiérrez-Martín, C. y Gómez-Limón, J.A. (2016). Water banks: What have we learnt from the international experience?, ***Water*** 8(10): 466. doi:10.3390/w8100466.
- Montilla-López, N.M., Gutiérrez-Martín, C. y Gómez-Limón, J.A. (2017). Impacto de la tarifación del agua de riego en el Bajo Guadalquivir, ***ITEA. Información Técnica Económica Agraria*** 113(1): 90-111. doi: 10.12706/itea.2017.006.

- Montilla-López, N.M., Gómez-Limón, J.A. y Gutiérrez-Martín, C. (2018). Sharing a river: Potential performance of a water bank for reallocating irrigation water, ***Agricultural Water Management*** 200: 47-59. doi: 10.1016/j.agwat.2017.12.025.

Capítulos de libro (con revisión por pares en editoriales de prestigio):

- Montilla-López, N.M., Gómez-Limón, J.A. y Gutiérrez-Martín, C. (2018). Simulating farmers' decision-making with a Cobb-Douglass MAUF. An application for an ex-ante policy analysis of water pricing. En Berbel, J., Bournaris, T., Manos, B., Matsatsinis, N. y Viaggi, D. (eds), *Multicriteria Analysis in Agriculture*. Springer, Dordrecht (The Netherlands).

Otros trabajos fruto de su investigación doctoral han sido igualmente publicados en diversos medios científicos o presentados en reuniones académicas, tal y como a continuación se relacionan:

Artículos científicos en revistas indexadas en otras bases de datos bibliográficas:

- Montilla-López, N.M., Gómez-Limón, J.A. y Gutiérrez-Martín, C. (2017). Los bancos de agua como instrumento económico para la mejora de la gestión del agua en España, ***Revista Española de Estudios Agrosociales y Pesqueros*** 247: 95-135.

Artículos en revistas de divulgación:

- Montilla-López, N.M., Gómez-Limón, J.A. y Gutiérrez-Martín, C. (2016). Los bancos de agua y su uso en España, ***Tierras de Castilla y León: Agricultura*** 244: 101-108.

Comunicaciones en congresos:

- Montilla-López, N.M. y Gutiérrez-Martín, C. (2015). *Impacto de la tarifación del agua de riego en el Bajo Guadalquivir tras la Reforma de la PAC*. Comunicación presentada en el X Congreso de la Asociación Española de Economía Agraria. Alimentación y territorios sostenibles desde el sur de Europa, 9-11 septiembre, Córdoba.
- Montilla-López, N.M.; Gómez-Limón, J.A. y Gutiérrez-Martín, C. (2016). *Retos y oportunidades de la implantación de los bancos de agua en España*. Comunicación presentada en el XXXIV Congreso Nacional de riegos, 7-9 junio, Sevilla.
- Montilla-López, N.M. (2016). *Los bancos de agua como instrumento para la mejora en el uso de los recursos hídricos en España*. Comunicación presentada en el V Congreso científico de investigadores en formación. Creando redes, 30 noviembre - 1 diciembre, Córdoba.
- Montilla-López, N.M., Gutiérrez-Martín, C. y Gómez-Limón, J.A. (2017). *Simulating farmers' decision-making with a Cobb-Douglas MAUF. An application for ex-ante policy analysis of water pricing*. Comunicación presentada en el XI Congreso Nacional de Economía Agraria. Sistemas

alimentarios y cambio global desde el Mediterráneo, 13-15 septiembre, Orihuela y Elche.

- Montilla-López, N.M., Gutiérrez-Martín, C. y Gómez-Limón, J.A. (2017). *Simulación del desempeño potencial de los bancos de agua: El caso de la Cuenca del Guadalquivir*. Comunicación presentada en el XI Congreso Nacional de Economía Agraria. Sistemas alimentarios y cambio global desde el Mediterráneo, 13-15 septiembre, Orihuela y Elche.
- Montilla-López, N.M. (2018). *¿Funcionarían los bancos de agua de reasignación de recursos hídricos en España?* Comunicación presentada en el VI Congreso científico de investigadores en formación. La generación del conocimiento, 18- 19 enero, Córdoba.

Estancia

La autora de la presente tesis doctoral ha realizado una estancia de investigación en el *Water Science Institute* en la Universidad Cranfield (Reino Unido), desde el 17/09/2017 al 19/12/2017, lo que le avala para solicitar la mención de "Doctor internacional".

Premios

Premio a la mejor comunicación oral en el área de Ciencias Sociales y Jurídicas obtenido en el VI Congreso científico de investigadores en formación. La generación del conocimiento, celebrado los días 18 y 19 de enero de 2018 en Córdoba.

Consideramos que a lo largo de su formación previa y en el periodo concreto de la tesis ha alcanzado la suficiente madurez científica, lo que le ha permitido obtener resultados en su investigación con una alta calidad contrastable internacionalmente, tal y como lo avalan los trabajos aceptados para su publicación.

Por todo ello, se autoriza la presentación de la tesis doctoral.

Córdoba, 23 de enero de 2018

Firma de los directores

Fdo.: José A. Gómez-Limón Rodríguez

Fdo.: Carlos Gutiérrez Martín

A mi familia,

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Llegó el momento de cerrar una nueva etapa de mi vida, y esta puede ser la primera vez que no temo enfrentarme a una página en blanco. Esta etapa ha sido complicada, tanto personal como académicamente, pero la concluyo con balance positivo. Siempre he pensado que de cada una de las personas que se cruzan en tu vida se aprende algo, y este es el motivo por el que quiero agradecer a todas aquellas personas que han estado cerca de mí, no solo en persona sino en la distancia.

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Resumen

La agricultura de regadío es mucho más rentable que la de secano, especialmente en zonas áridas o semiáridas, donde los usos agrícolas representan hasta el 80% de los usos del agua. Sin embargo, la agricultura se enfrenta a uno de los mayores problemas que existen en la actualidad: la creciente escasez de agua. Esto está produciendo el "cierre" de numerosas cuencas, situación donde no existen posibilidades de incrementar la oferta y los recursos disponibles están asignados. Por tanto, la demanda de cualquier nuevo usuario potencial (incluido el medio ambiente) no puede satisfacerse sin disminuir la cantidad asignada a otros usuarios.

En estas circunstancias, es necesario implementar instrumentos económicos de gestión de la demanda, que doten al sistema de asignación de derechos de la flexibilidad necesaria para que los recursos existentes puedan reasignarse de forma dinámica hacia los usos con mayor demanda social existente en cada momento. Entre dichos instrumentos destacan la tarifación del agua y los mercados y bancos de agua.

A pesar de que la Directiva Marco de Agua (DMA) considera la *tarifación del agua* como un elemento clave, los estudios empíricos desarrollados en esta tesis en dos zonas regables en la cuenca del Guadalquivir refutan la utilidad de este instrumento para la reasignación de recursos. Efectivamente, el consumo de agua no disminuye cuando se aplican tarifas para recuperar los costes de agua (menores a 0,15-0,20 €/m^3), ya que para este rango de precios la demanda de agua resulta inelástica, dado que los agricultores no cambian el plan de cultivos hasta que el precio del agua es mucho más elevado (superiores a 0,20-0,30 €/m^3). Por tanto, la implementación de la tarifación como sugiere la DMA no produciría reasignaciones del recurso que permitiesen una mejora en la eficiencia del uso del agua. Por el contrario, dicha implementación solo produciría transferencias desde el sector agrario al público, resultando en una importante pérdida de rentas agrarias. Si la tarifación alcanzase el tramo elástico (por encima de lo exigido por la DMA), sí se produciría ahorro de agua, pero a costa de pérdidas significativas de eficiencia económica y de empleo, lo que pondría en riesgo aquellas zonas donde la agricultura es una fuente importante de actividad económica.

Una extensa revisión de la literatura sobre las experiencias de los *bancos de agua*, tanto a nivel nacional como internacional, ha evidenciado la utilidad potencial de este instrumento para la gestión de los recursos hídricos en cuencas "cerradas", como herramienta capaz de minimizar los impactos negativos de la escasez, tanto estructural como coyuntural. Al igual que el resto de los mercados de agua, los bancos de agua posibilitan que los recursos se reasignen de forma voluntaria hacia los usos de mayor valor, pero estos últimos cuentan con la ventaja adicional de reducir los costes de transacción estáticos asociados a las operaciones de intercambio, y la posibilidad de un mejor control público de las transacciones.

El análisis ha revelado igualmente que los bancos de agua son instrumentos económicos muy flexibles, en la medida que pueden adoptar diversos diseños, cada de uno de los cuales con sus propias ventajas e inconvenientes. En este sentido, la experiencia internacional evidencia que los bancos públicos de agua activos para la reasignación temporal de derechos resultan ser una herramienta útil para minimizar los efectos de la escasez coyuntural del agua (gestión de sequías). La implementación de este tipo de banco de agua para el caso de la cuenca del Guadalquivir se ha simulado mediante programación matemática. Esta simulación ha permitido cuantificar *ex-ante* los impactos del instrumento para distintos escenarios de disponibilidad de agua. Los resultados muestran que este tipo de banco resultaría muy útil para la gestión de las sequías a las que recurrentemente se ve sometida esta cuenca, ya que facilitaría transferencias temporales de los recursos escasos desde usos de menor valor a otros de mayor valor añadido (minimización del impacto económico), evitando con ello reducciones severas en el empleo generado por el regadío (minimización del impacto social).

Finalmente, para las simulaciones realizadas de ambos instrumentos económicos, se ha desarrollado un nuevo enfoque metodológico que modeliza la toma de decisiones del agricultor mediante una función multi-atributo del tipo Cobb-Douglas. El procedimiento de calibración propuesto resulta sencillo y permite generar resultados de simulación más cercanos a la realidad, por lo que resulta de gran interés para las simulaciones *ex-ante* de los impactos de los instrumentos económicos.

Abstract

Irrigated agriculture is much more profitable than rain-fed agriculture, especially in arid and semi-arid regions, where the consumption of this sector accounts for up to 80% of total water use. However, one paramount problem is jeopardizing the future of irrigated agriculture: the increasing water scarcity. This is leading to the closure of basins, a situation where it is not possible to further expand water supply and the currently available resources are fully allocated. Thus, any new potential water user (including the environment) cannot be satisfied without reducing the amount of water already allocated to other users.

Under these circumstances, it is necessary to implement demand-side economic instruments that make the water rights system more flexible, in order to allow a dynamic reallocation of water resources towards those with greater social demand. Among these water pricing, water markets and water banks are highlighted.

Although the Water Framework Directive (WFD) considers *water pricing* as a key element for the implementation of demand-side water policy, the empirical studies developed herein that focused on two irrigation districts located in the Guadalquivir River Basin (GRB) refute the usefulness of this economic instrument for reallocating water currently granted for irrigation purposes. In fact, it has been shown that irrigation water consumption does not decrease when cost-recovering tariffs (bellow 0.15-0.20 €/m^3) are implemented because water demand is inelastic over this price range; that is, farmers are not willing to change their cropping plans until the tariffs are much higher (above 0.20-0.30 €/m^3). Thus, the implementation of water pricing as the WFD suggests neither results in water resources reallocation nor water use efficiency improvement. By contrast, the implementation of this instrument at these prices only results in monetary transfers from the agricultural sector to the public sector, resulting in significant losses of farmers' income. If water tariffs reach the elastic segment of the demand (above the WFD requirements), this would lead not only to water savings but also to losses of economic efficiency and agricultural employment, which would be significant, thus jeopardizing rural development in regions where agriculture is an important economic activity.

An extensive literature review regarding the experience in implementing *water banks*, both at international and national levels, has demonstrated the potential of this instrument for the management of water resource in a ”closed basin” as a suitable tool to minimize the negative impacts of both structural and cyclical scarcity. Similar to other water markets, water banks permit the voluntary reallocation of water resources towards high-value uses, with the additional advantage of reducing the static transaction costs involved and more effective public control over transactions.

This analysis has also revealed that water banks are flexible instruments because they can adopt several designs, each with its own advantages and disadvantages. In this sense, international experience demonstrates that a publicly run active water bank operating at basin level designed to temporarily reallocate water resources is a useful instrument to minimize the effects of cyclical scarcity (drought management). We, therefore, performed an empirical analysis to simulate the implementation of this type of water bank within the GRB using mathematical programming models. This simulation exercise has allowed an *ex-ante* assessment of the impacts of this economic instrument under different water availability scenarios. The results obtained confirm that this type of water bank is suitable for the management of droughts faced cyclically in this basin because it facilitates temporary water transfer from low to high value-added uses (minimization of economic impact), while simultaneously avoiding reductions in the employment generated by irrigated agriculture (minimization of social impact).

Capítulo 1
Introducción y objetivos

1.1. Introducción

1.1.1. La escasez del agua

Desde hace algunas décadas, se está observando que la explotación antrópica de los recursos hídricos en algunas zonas del planeta está llegando a sus límites de sostenibilidad, dada la imposibilidad de que la oferta se incremente al mismo ritmo que lo hace la demanda. El aumento de la población y la consecuente demanda de alimentos ha provocado que a lo largo del siglo XX la superficie de regadío se incremente considerablemente a nivel global, convirtiéndose en la actividad humana con mayores extracciones y consumo de agua a nivel mundial (FAO, 2012). Esta presión sobre los recursos hídricos es especialmente intensa en zonas semiáridas como California, Australia o España, donde la agricultura de regadío resulta especialmente competitiva, y los usos agrícolas representan hasta el 80% de los usos del agua. Además, esta situación de escasez relativa del recurso se prevé se acentúe en un futuro próximo como consecuencia del calentamiento global y el cambio climático provocado por este, tanto de forma estructural (a largo plazo) por el incremento de las necesidades hídricas de los cultivos y el descenso de la oferta de agua por la disminución de las precipitaciones, como de forma coyuntural (cíclicamente a corto plazo) por la mayor frecuencia e intensidad de los períodos de sequía en todo el planeta (IPCC, 2014).

En muchas de estas cuencas hidrográficas no es posible aumentar la oferta de agua porque las localizaciones idóneas en las que construir nuevos embalses y demás infraestructura asociada ya están ocupadas o tienen un coste desproporcionado, el agua subterránea está sobreexplotada, y otras fuentes alternativas de agua (p. ej., agua desalada) tienen un coste que no puede asumir el agricultor medio. En este sentido, ante la imposibilidad de atender nuevas demandas, se está produciendo el "cierre de cuencas" (Falkenmark y Molden, 2008; Molle *et al.*, 2010), término con el que se conoce a la situación en la que se alcanza el techo de oferta y, por tanto, las nuevas demandas solo pueden atenderse a costa de reducir la disponibilidad de agua de otros usuarios. En España, esto ha ocurrido principalmente de las cuencas del sur y el este peninsular (Berbel *et al.*, 2013), como es el caso de la cuenca del río Guadalquivir, que será utilizada como caso de estudio en esta investigación.

Cabe afirmar que el cierre de las cuencas se corresponde con la última fase de la denominada "madurez de la economía del agua" (Randall, 1981), que se caracteriza principalmente por: i) una oferta inelástica del recurso en el largo plazo; ii) una alta y creciente demanda del agua; iii) una competencia cada vez más intensa por el uso del agua entre los distintos sectores económicos (agricultura, industria, producción de energía y ocio), urbanos y el medio ambiente (mantenimiento de los caudales ecológicos); iv) externalidades ambientales negativas; y v) un coste de suministro del recurso cada vez mayor, debido a las crecientes inversiones necesarias para mantener en buen estado las infraestructuras hidráulicas existentes (embalses, sistemas de distribución) y el elevado coste de las nuevas fuentes de agua (desalación, reutilización, etc.) que han tenido que desarrollarse ante la imposibilidad de contar con más recursos convencionales (Gómez-Limón y Calatrava, 2016).

1.1.2. Política del agua para la gestión de la escasez: los instrumentos económicos

Bajo las condiciones anteriormente expuestas, resulta evidente la necesidad de promover una gestión más eficiente del agua actualmente disponible mediante "políticas de demanda", que permitan una asignación preferente del recurso hacia usos que generen mayor valor económico para el conjunto de la sociedad, ya sean estos el abastecimiento de la población, la realización de actividades económicas (agricultura, industria, energía), la sostenibilidad ambiental (buen estado de las masas de agua) o la provisión de bienes públicos como el paisaje y actividades recreativas (baño, pesca, etc.). Dentro de esta política, la agricultura de regadío tendrá un papel central como principal usuario actual del recurso.

En enero de 1992, las Naciones Unidas, en la Declaración de Dublín sobre el Agua y el Desarrollo Sostenible de la Conferencia Internacional del Agua y el Medio Ambiente, reconocieron el agua como un *bien económico*. Hoy día está ampliamente aceptado que la gestión de los recursos hídricos por su valor económico es "una importante forma de lograr un uso eficiente y equitativo, y de fomentar la conservación y protección de los recursos hídricos". Esta Declaración confirma la importancia del agua como recurso escaso, tanto en cantidad como calidad y accesibilidad, y las reglas que deben ser establecidas en cuanto a su gestión y gobernanza (Berbel *et al.*, 2017).

En la actualidad, los regímenes de asignación de derechos sobre los recursos hídricos suelen basarse en criterios históricos, siendo normalmente muy poco flexibles, lo que impide una rápida redistribución de los mismos en función de las cambiantes demandas sociales y coyuntura de los mercados. Esto ha llevado a que todos los gobiernos responsables de cuencas cerradas hayan hecho intentos de mejorar la gestión de los recursos hídricos mediante instrumentos de demanda (reasignación de derechos). No obstante, la puesta en marcha de estos mecanismos requiere de una fuerte voluntad política capaz de superar las reticencias de los agentes afectados, y dotar a las administraciones responsables de la gestión pública del agua de los medios humanos y económicos para la implementación de las medidas necesarias, circunstancias que no suelen darse hasta que la escasez del agua se percibe de forma severa (Rey *et al.*, 2018). En cualquier caso, existe ya un número creciente de países que se han visto abocados a reformar la legislación que regula la asignación de agua, haciendo que esta sea más flexible mediante el uso de *instrumentos económicos* (OCDE, 2015). Se evidencia así la importancia de la implementación de este tipo de instrumentos, tales como la tarifación del agua y los mercados y bancos de agua, que doten a las administraciones responsables de su gestión de mecanismos para minimizar los efectos de la creciente de escasez (estructural y coyuntural) del agua mediante la reasignación y el uso eficiente de los recursos existentes (Dinar *et al.*, 1997; Sumpsi *et al.*, 1998; Lago *et al.*, 2015).

En Europa, la Directiva Marco de Agua (DMA) destaca el uso de la *tarifación del agua* como principal instrumento económico para la implementación de la política de demanda de agua. En última instancia este instrumento pretende desincentivar el uso del recurso en actividades económicas poco rentables, favoreciendo así una asignación más eficiente y racional del agua entre sus potenciales usuarios (Lee y Jouravlev, 1998). Además, la tarifación propuesta por la DMA persigue incrementar la capacidad de recuperación de los costes derivados de los servicios del agua. Esta apuesta por la tarifación como instrumento económico común para la gestión del agua en la Unión Europea ha sido ratificada por la Comisión Europea (2012) en el documento "Plan para salvaguardar los recursos hídricos de Europa" (*Blueprint to Safeguard Europe's Water Resources*), donde se señala que la inadecuada implementación de la tarifación del agua es la responsable de mal estado de las masas de agua en el continente. Así, este mismo documento apunta la necesidad de una aplicación más estricta de este instrumento económico de cara a mejorar la gestión del recurso.

En España, la implementación de la tarifación del agua no se ha realizado de forma estricta conforme a lo establecido en la DMA, en la medida que las administraciones responsables de la gestión del recurso sólo recuperan parcialmente los costes de los servicios prestados a los agentes económicos que usan el agua. Esta aplicación parcial de la tarifación, especialmente en el sector agrario, está justificada por un gran número de trabajos empíricos *ex ante* que han analizado los efectos de la tarifación del agua (Gómez-Limón y Riesgo, 2004; Mejías *et al.*, 2004; Iglesias y Blanco, 2008), donde se pone de manifiesto que este instrumento no es tan efectivo como cabría esperar en zonas con escasez estructural, en las cuales la tarifación no genera ahorros de agua, sino únicamente transferencia de rentas de los agricultores al sector público (Calatrava *et al.*, 2011; Kahil *et al.*, 2016).

Por su parte, países como Estados Unidos o Australia han apostado por los *mercados y bancos de agua* como principal instrumento para flexibilizar el sistema de asignación de recursos hídricos entre sus potenciales usuarios y, con ello, mejorar la eficiencia en los diferentes usos del agua (Easter y Huang, 2014). De esta manera, los mercados y bancos de agua han permitido favorecer una reasignación de derechos de uso de agua en aquellas zonas donde los recursos disponibles ya están asignados y existen usuarios que demandan una mayor cantidad de agua, incluido el medio ambiente (Wheeler *et al.*, 2013; Pérez-Blanco y Gutiérrez-Martín, 2017). Así, aunque los volúmenes de agua transaccionadas han sido en general escasos (Brewer *et al.*, 2008), los mercados y bancos de agua han sido implementados con éxito en estos países.

En España también se han implementado los mercados y los bancos de agua como complemento a la tarifación del recurso exigido por la DMA, si bien los resultados hasta la fecha han sido más limitados que en los países de tradición anglosajona. Así, cabe comentar que la actividad del conjunto del mercado en España ha sido relativamente escasa, y sólo se han producido operaciones en periodos de sequía. De hecho, en el año de mayor actividad del mercado (año 2007) el volumen de agua intercambiado no llegó a suponer ni tan siquiera el 0,5% del total del agua usada a nivel nacional (Palomo-Hierro *et al.*, 2015). Del total de operaciones, la mayoría se realizaron mediante mercados de derechos, y tan sólo una cuarta parte se realizaron a través de centros de intercambio (bancos de agua). Dentro de esta última figura cabe señalar la implementación de bancos de agua durante el período de sequía 2006-2008 en la cuenca del Júcar, donde se movilizaron casi el 1% de los recursos usados en la misma, y en las cuencas del Guadiana y del Segura, donde la actividad de las operaciones de mercado fue menor, movilizando menos del 0,5% de los recursos usados (Palomo-Hierro *et al.*, 2015). Estos datos ponen de manifiesto que, hasta la fecha, estos instrumentos basados en el intercambio de derechos de agua se han implementado de forma timorata, y su contribución a la mejora de gestión del agua ha sido limitada, muy por debajo del su desempeño potencial.

1.2. Retos de investigación: hipótesis y objetivos

La presente investigación surge con el objetivo de intentar dar respuesta a algunas lagunas de conocimiento existentes en relación con la aplicación de instrumentos económicos, tanto desde una perspectiva metodológica como empírica. En última instancia, el nuevo conocimiento generado por esta investigación busca servir de apoyo para la toma de decisiones encaminadas a la adopción de los cambios normativos e institucionales necesarios en España para optimizar el diseño e implementación de estos instrumentos, posibilitando con ello una mejora efectiva de la gestión pública del agua.

En este sentido, la **hipótesis de partida** en la que se sustentan los objetivos de esta tesis doctoral es que la implementación de instrumentos económicos resulta adecuada para la mejora de la gestión de los recursos hídricos en España, en la medida que puede contribuir a lograr una buena gobernanza del agua, permitiendo asimismo alcanzar un equilibrio socialmente aceptable entre eficiencia económica y sostenibilidad ambiental. De manera más concreta, los instrumentos económicos objeto de estudio son la *tarifación del agua* y los *bancos de agua*, que a priori cabe presuponer como los más adecuados para la consecución de los objetivos públicos de la política hídrica. Ambas herramientas tienen en común que tratan de mejorar la eficiencia económica del recurso haciendo que el agua se utilice en aquellas actividades que generan mayor valor añadido, contribuyendo así a mejorar el bienestar del conjunto de la sociedad.

Para ello, se propone como **objetivo principal** de la presente investigación analizar el desempeño potencial de distintos instrumentos económicos para la mejora en la gestión de los recursos hídricos en España. Este objetivo se pretende alcanzar tanto desde una perspectiva teórica, realizando una extensa revisión de literatura que incluya las ventajas y los inconvenientes de estos instrumentos, como desde una perspectiva empírica, mediante la simulación de su implementación mediante programación matemática, al objeto de analizar los impactos, tanto positivos como negativos, que tendría la implementación práctica de estas herramientas en el caso de estudio de la cuenca del río Guadalquivir.

Para alcanzar este objetivo principal se hace necesario abordar una serie **objetivos específicos**, que podemos clasificar en: i) *empíricos*, aquellos que tienen relación con la observación de la realidad pasada y con la simulación de modelos; y ii) *metodológicos*, aquellos centrados en el análisis y desarrollo de metodologías para alcanzar los objetivos empíricos. A continuación, se describen dichos objetivos específicos, y se indica en qué capítulos (cada uno correspondiente a una publicación) se abordan.

I. Objetivos específicos de carácter empírico:

I.1.- Simular el desempeño de la tarifación volumétrica del agua de riego y analizar los impactos económicos, sociales y medioambientales de este instrumento económico. Para ello se han realizado los trabajos de investigación que se corresponden con el artículo de Montilla-López *et al.* (2017) y el capítulo de libro de Montilla-López *et al.* (2018b), y que se recogen en el Capítulo 2 y el Capítulo 3 de esta tesis doctoral, respectivamente.

I.2.- Analizar críticamente el desempeño de los bancos de agua a nivel internacional mediante la revisión de evidencias procedentes de diferentes casos de estudios recogidos en la literatura, al objeto de poder establecer las ventajas e inconvenientes de este instrumento económico. Este objetivo se ha trabajado en el artículo de Montilla-López *et al.* (2016), que corresponde con el Capítulo 4 de esta tesis doctoral.

I.3.- Simular el funcionamiento de un banco de agua activo para la reasignación de recursos hídricos entre los agricultores y analizar los impactos económicos, sociales y medioambientales de este instrumento económico. Este objetivo se ha desarrollado en el artículo de Montilla-López *et al.* (2018a), que se corresponde con Capítulo 5 de esta tesis doctoral.

II. Objetivos específicos de carácter metodológico:

II.1.- Realizar un análisis comparativo de diferentes métodos de calibración de la función objetivo de los modelos de programación matemática que simulan el comportamiento de los agricultores, al objeto de determinar el enfoque metodológico que permite *generar resultados de simulación más precisos.* Este objetivo se ha alcanzado a través del trabajo recogido en el artículo de Montilla-López *et al.* (2017), que se incluye en el Capítulo 2 de esta tesis doctoral.

II.2.- Desarrollar un nuevo método de calibración de una función de utilidad multi-atributo tipo Cobb-Douglas, como innovación metodológica con la que se pretende realizar simulaciones del comportamiento de los agricultores más robustas. Esta investigación se ha llevado a mediante el mencionado capítulo de libro de Montilla-López *et al.* (2018b), que se recoge en el Capítulo 3 de esta tesis doctoral.

La consecución de los objetivos específicos anteriores resulta de especial interés para los decisores políticos responsables de la toma de decisión relacionadas con la gestión pública de los recursos hídricos. Efectivamente, los resultados derivados de los objetivos específicos de carácter empírico suponen un soporte a la toma decisiones políticas en relación con la selección y diseño de los instrumentos económicos a implementar para mejorar la gestión del agua desde la perspectiva de la demanda. Asimismo, la consecución de los objetivos específicos de carácter metodológico es de gran utilidad para los investigadores y técnicos encargados de la evaluación *ex-ante* de los previsibles impactos de los instrumentos económicos analizados, especialmente por el desarrollo de un nuevo método de calibración de funciones multi-atributo Cobb-Douglas para la simulación del comportamiento de los regantes, que se ha demostrado ser más preciso que otros métodos que le preceden.

1.3. Estructura del documento

Dentro de este contexto, para afrontar cada uno de los objetivos propuestos, el presente trabajo se plantea de la siguiente manera. Tras este capítulo introductorio, el **Capítulo 2**, correspondiente al artículo de Montilla-López *et al.* (2017), se centra en el análisis del efecto de la tarifación sobre la demanda de agua en sistemas de regadío, así como en los impactos, tanto económicos como sociales, que tendría su aplicación tras la reforma de la Política Agraria Común (PAC) en la Comunidad de Regantes del Sector BXII del Bajo Guadalquivir. Para ello se plantea un trabajo empírico basado en modelos de programación matemática, realizando una comparativa crítica de los resultados obtenidos mediante tres conocidos métodos de programación, como son: a) la clásica maximización del beneficio; b) la Programación Matemática Positiva propuesta formalmente por Howitt (1995); y c) la Programación por Metas Ponderadas propuesta por Sumpsi *et al.* (1997).

El **Capítulo 3** es una reproducción del capítulo de libro de Montilla-López *et al.* (2018b), donde se desarrolla un nuevo enfoque para simular la toma de decisiones del agricultor mediante la programación matemática. Más concretamente, se propone un nuevo método de calibración de los parámetros de una función objetivo multi-atributo del tipo Cobb-Douglas, al objeto de dotar de mayor realismo a la simulación de la toma de decisiones de estos agentes económicos. Dicho método de calibración está basado en la Teoría de la Utilidad Multi-Atributo, y trata de superar algunas de las deficiencias que caracterizan los enfoques empleados hasta la fecha basados en funciones de utilidad aditiva y resulta más sencillo que otros métodos de calibración de funciones de utilidad multi-atributo Cobb-Douglas (Gutiérrez-Martín y Gómez-Gómez, 2011; Gómez-Limón *et al.*, 2016). Este método propuesto basado en el uso de funciones de utilidad multi-atributo tipo Cobb-Douglas es comparado, con fines ilustrativos, con la maximización del beneficio y con la maximización de una función de utilidad multi-atributo aditiva calibrada mediante Programación por Metas Ponderadas. Dicho análisis comparativo se realiza aplicando estos tres enfoques a un caso de estudio empírico, que trata de simular los impactos de la tarifación volumétrica del agua de riego en la Comunidad de Regantes del Canal de la Margen Izquierda del Bembézar. De esta manera, se evidencian las ventajas de simulación que emplea el método de calibración propuesta para una función de utilidad Cobb-Douglas. Por tanto, este capítulo tiene un doble propósito, uno de carácter empírico orientado analizar los efectos de la tarifación del agua de riego, y otro de índole metodológico centrado en el desarrollo de un nuevo método de simulación, que será utilizado posteriormente en los modelos construidos para el Capítulo 5.

En el **Capítulo 4**, reproducción fiel del artículo Montilla-López *et al.* (2016), se estudia la potencialidad de los bancos de agua (denominados *centros de intercambio* en la legislación española) como instrumento económico de gestión de la demanda enfocada hacia la satisfacción de las necesidades de agua en un contexto de cuencas cerradas (imposibilidad de aumento de la oferta de agua) y fuerte incertidumbre en cuanto la disponibilidad del recurso (impacto creciente de las sequías como consecuencia del cambio climático). Con este objetivo, se realiza un análisis crítico de la implementación real de los bancos de agua a nivel nacional e internacional, al objeto de poder analizar las ventajas e inconvenientes de este instrumento económico para la gestión de la escasez e incertidumbre de agua dentro de la política hidráulica. De dicho análisis surgen una serie de recomendaciones para la mejora del diseño de este instrumento económico dentro de la política del agua en España, orientadas a la implementación más eficiente y eficaz de los bancos de agua en nuestro país.

El **Capítulo 5** se corresponde con el artículo de Montilla-López *et al.* (2018a), en el que se lleva a cabo una simulación de un banco de agua activo para la reasignación temporal de derechos de agua entre agricultores de regadío en un contexto de escasez. La propuesta de tipo de banco de agua para su estudio empírico está fundamentada en los resultados del Capítulo 4, dada la gran potencialidad de este tipo de banco para mejorar la eficiencia en el uso de agua en cuencas cerradas como la del Guadalquivir. Este trabajo empírico se basa en modelos de simulación de programación matemática desarrollado con la metodología propuesta en el Capítulo 3 para calibrar los parámetros de la función de utilidad multi-atributo Cobb-Douglas. De esta manera, se simula la reasignación del agua resultante para diferentes escenarios de sequía y costes de transacción, así como sus efectos económicos y sociales.

Finalmente, el **Capítulo 6** muestra las conclusiones derivadas del conjunto de la investigación y sus implicaciones para la gestión de los recursos hídricos en España. En este sentido, se aportan algunas sugerencias de mejora de la legislación nacional en materia de instrumentos de demanda.

A modo de resumen, la Tabla 1.1 muestra la correspondencia entre los capítulos de la presente tesis doctoral, las publicaciones a las que está referida y la consecución de los objetivos específicos alcanzados en cada uno de ellos.

Igualmente, en relación con la estructura de este documento conviene señalar que, en aras del cumplimiento del artículo 35 de la Normativa Reguladora de los Estudios de Doctorado propuesta por la Comisión de Másteres y Doctorado de la Universidad de Córdoba, para la obtención del título de doctor con "*Mención Internacional*", las conclusiones se encuentran redactadas en inglés. Adicionalmente, y para no alterar el texto original, también están redactadas en inglés los capítulos 3, 4 y 5.

Tabla 1.1. *Correspondencia entre capítulos, publicaciones y objetivos específicos de la tesis*

Capítulo	Referencia	Objetivo específico
2	Montilla-López, N.M., Gutiérrez-Martín, C. y Gómez-Limón, J.A. (2017). Impacto de la tarifación del agua de riego en el Bajo Guadalquivir, *ITEA. Información Técnica Económica Agraria* 113(1): 90-111.	I.1, II.1
3	Montilla-López, N.M., Gómez-Limón, J.A. y Gutiérrez-Martín, C. (En prensa). Simulating farmers' decision-making with a Cobb-Douglass MAUF. An application for an ex-ante policy analysis of water pricing. En Berbel, J., Bournaris, T., Manos, B., Matsatsinis, N. y Viaggi, D. (eds), *Multicriteria Analysis in Agriculture*. Springer, Dordrecht (The Netherlands).	I.1, II.1, II.2
4	Montilla-López, N.M., Gutiérrez-Martín, C. y Gómez-Limón, J.A. (2016). Water banks: What have we learnt from the international experience?, *Water* 8(10): 466.	I.2
5	Montilla-López, N.M., Gómez-Limón, J.A. y Gutiérrez-Martín, C. (En prensa). Sharing a river: Potential performance of a water bank for reallocating irrigation water, *Agricultural Water Management* 200: 47-59.	I.3

1.4. Publicaciones y actividades derivadas de la tesis

Además de las cuatro publicaciones principales que forman el cuerpo principal de la tesis, se han desarrollado otras publicaciones y actividades durante su realización. A continuación, se muestran todas las publicaciones que se han derivado de la investigación.

Artículos en revistas indexadas en el Journal Citation Report (JCR):

- Montilla-López, N.M., Gutiérrez-Martín, C. y Gómez-Limón, J.A. (2016). Water banks: What have we learnt from the international experience?, ***Water*** 8(10): 466. doi:10.3390/w8100466.
- Montilla-López, N.M., Gutiérrez-Martín, C. y Gómez-Limón, J.A. (2017). Impacto de la tarifación del agua de riego en el Bajo Guadalquivir, ***ITEA. Información Técnica Económica Agraria*** 113(1): 90-111. doi: 10.12706/itea.2017.006.
- Montilla-López, N.M., Gómez-Limón, J.A. y Gutiérrez-Martín, C. (2018). Sharing a river: Potential performance of a water bank for reallocating irrigation water, ***Agricultural Water Management*** 200: 47-59. doi: 10.1016/j.agwat.2017.12.025.

Capítulos de libro (con revisión por pares):

- Montilla-López, N.M., Gómez-Limón, J.A. y Gutiérrez-Martín, C. (2018). Simulating farmers' decision-making with a Cobb-Douglass MAUF. An application for an ex-ante policy analysis of water pricing. En Berbel, J., Bournaris, T., Manos, B., Matsatsinis, N. y Viaggi, D. (eds), *Multicriteria Analysis in Agriculture*. Springer, Dordrecht (The Netherlands).

Artículos científicos en revistas indexadas en otras bases de datos bibliográficas (con revisión por pares):

- Montilla-López, N.M., Gómez-Limón, J.A. y Gutiérrez-Martín, C. (2017). Los bancos de agua como instrumento económico para la mejora de la gestión del agua en España, ***Revista Española de Estudios Agrosociales y Pesqueros*** 247: 95-135.

Artículos en revistas de divulgación:

- Montilla-López, N.M., Gómez-Limón, J.A. y Gutiérrez-Martín, C. (2016). Los bancos de agua y su uso en España, ***Tierras de Castilla y León: Agricultura*** 244: 101-108.

Comunicaciones en congresos:

- Montilla-López, N.M. y Gutiérrez-Martín, C. (2015). *Impacto de la tarifación del agua de riego en el Bajo Guadalquivir tras la Reforma de la PAC*. Comunicación presentada en el X Congreso de la Asociación Española de Economía Agraria. Alimentación y territorios sostenibles desde el sur de Europa, 9-11 septiembre, Córdoba.
- Montilla-López, N.M.; Gómez-Limón, J.A. y Gutiérrez-Martín, C. (2015). *Retos y oportunidades de la implantación de los bancos de agua en España*. Comunicación presentada en el XXXIV Congreso Nacional de riegos, 7-9 junio, Sevilla.
- Montilla-López, N.M. (2016). *Los bancos de agua como instrumento para la mejora en el uso de los recursos hídricos en España*. Comunicación presentada en el V Congreso científico de investigadores en formación. *Creando redes*, 30 noviembre - 1 diciembre, Córdoba.
- Montilla-López, N.M., Gutiérrez-Martín, C. y Gómez-Limón, J.A. (2017). *Simulating farmers' decision-making with a Cobb-Douglas MAUF. An*

application for ex-ante policy analysis of water pricing. Comunicación presentada en el XI Congreso Nacional de Economía Agraria. Sistemas alimentarios y cambio global desde el Mediterráneo, 13-15 septiembre, Orihuela y Elche.

- Montilla-López, N.M., Gutiérrez-Martín, C. y Gómez-Limón, J.A. (2017). *Simulación del desempeño potencial de los bancos de agua: El caso de la Cuenca del Guadalquivir*. Comunicación presentada en el XI Congreso Nacional de Economía Agraria. Sistemas alimentarios y cambio global desde el Mediterráneo, 13-15 septiembre, Orihuela y Elche.
- Montilla-López, N.M. (2018). *¿Funcionarían los bancos de agua de reasignación de recursos hídricos en España?* Comunicación presentada en el VI Congreso científico de investigadores en formación. La generación del conocimiento, 18-19 enero, Córdoba.

Asimismo, debe indicarse que, durante el desarrollo de la tesis, la doctoranda ha realizado una **estancia en el extranjero** de tres meses en el *Water Science Institute* en la Universidad de Cranfield (Reino Unido).

Finalmente, como indicador adicional de la calidad de la investigación doctoral realizada, se señala que la doctoranda ha sido galardonada con el **premio a la mejor comunicación oral** en el área de Ciencias Sociales y Jurídicas en el *VI Congreso científico de investigadores en formación*, celebrado los días 18 y 19 de enero de 2018 en Córdoba.

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Capítulo 2
Impacto de la tarifación del agua de riego en el Bajo Guadalquivir

Impacto de la tarifación del agua de riego en el Bajo Guadalquivir[1]

Resumen

Este trabajo se centra en el análisis del efecto de la tarifación sobre la demanda de agua en sistemas de regadío intensivos, así como los impactos tanto económicos como sociales que tendría su aplicación tras la reforma de la Política Agraria Común recientemente aprobada. La zona regable considerada como caso de estudio ha sido la Comunidad de Regantes del Sector BXII del Bajo Guadalquivir. La unidad de análisis ha sido la explotación agraria, distinguiéndose tres explotaciones tipo características de la zona de estudio. Para cada una de estas explotaciones tipo se han construido tres modelos de simulación, basados en tres métodos de programación matemática diferenciados. Con ellos se ha podido simular de forma más robusta la toma de decisiones de los regantes. A partir de tales simulaciones se han podido cuantificar los previsibles impactos de este instrumento sobre el consumo de agua, las rentas agrarias, la recaudación pública, la generación de empleo y el bienestar social asociado al uso agrario del agua. Los resultados obtenidos para las diferentes metodologías aplicadas han resultado ser muy similares. El análisis de las curvas de demanda muestra un primer tramo inelástico hasta precios elevados (0,3 €/m^3), seguidos de tramos elásticos donde los regantes realizarían cambios significativos en sus planes de cultivo. En cuanto al análisis de impactos socioeconómicos de la tarifación, los resultados muestran que en ambos tramos se producen pérdidas de renta agraria. Sin embargo, en el tramo elástico estas pérdidas de renta son superiores a la recaudación por tarifación, lo que provoca importantes pérdidas de eficiencia económica y de empleo generado.

Palabras clave: Regadío, instrumentos económicos, demanda de agua, programación matemática, impacto socioeconómico.

[1] El contenido de este capítulo coincide con el del artículo siguiente:
Montilla-López, N.M., Gutiérrez-Martín, C. y Gómez-Limón, J.A. (2017). Impacto de la tarifación del agua de riego en el Bajo Guadalquivir, *ITEA. Información Técnica Económica Agraria* 113(1): 90-111.

2.1. Introducción

La escasez estructural de recursos hídricos y su incierta variabilidad espacial y temporal se han convertido actualmente en uno de los principales problemas al que debe enfrentarse la agricultura española. Además, esta problemática situación es probable que empeore en un futuro próximo como consecuencia de la gran vulnerabilidad de España al impacto del cambio climático. Dada la demanda creciente del recurso y la dificultad (elevado coste) de disponer de nuevos recursos, se ha hecho necesario un cambio en la forma de gestión de los recursos hídricos, pasando del tradicional enfoque de "políticas de oferta" a un enfoque de "políticas de demanda", orientadas a la mejora de la eficiencia en el uso del agua y la compatibilidad de su uso por parte de las actividades económicas y la sostenibilidad ambiental (Calatrava *et al*., 2015).

En este contexto no debe obviarse, sin embargo, que la puesta en riego ha sido una de las estrategias más efectivas de desarrollo rural implementada durante el último siglo. Efectivamente, la transformación en regadío ha supuesto una mejora generalizada, de la rentabilidad del sector agrario, principalmente por el incremento y estabilización de las producciones y, por tanto, de las rentas agrarias. Igualmente, el incremento del rendimiento de los cultivos, así como la posibilidad de desarrollar otros cultivos inviables en condiciones de secano, hace que la agricultura de regadío pueda contribuir de manera más eficiente al bienestar del conjunto de la sociedad, aumentando la cantidad de alimentos y su variedad, así como generando un impacto social positivo en el medio rural donde se localiza, contribuyendo a la fijación de población en el territorio y la viabilidad de las comunidades rurales.

A pesar de las ventajas del regadío antes comentadas, también es cierto que este tipo de agricultura es la actividad con mayor consumo de recursos hídricos. Un ejemplo de ello puede encontrarse en las cuencas del sur de Europa, donde el desarrollo de un importante sector de regadío ha provocado el "cierre" de las mismas. La cuenca del Guadalquivir puede considerarse un caso paradigmático de este proceso, donde se estima que en el año 2015 el uso agrario represente aproximadamente el 88% del total del agua, frente al 10% del uso urbano.

En este sentido, el Parlamento Europeo y el Consejo Europeo de Ministros aprobaron la Directiva Marco del Agua (DMA) en el año 2000, principal norma del marco comunitario de actuación en política de aguas. Esta Directiva exige a los Estados miembros realizar análisis económicos del uso del agua, aplicar el principio de recuperación de los costes de los servicios relacionados con el agua, así como el cumplimiento de objetivos medioambientales, con el fin de alcanzar el buen estado de las masas de agua, cumpliendo el principio de "quien contamina paga". Con ello se trata de conseguir una gestión sostenible de los recursos hídricos y hacer frente a la creciente presión en la demanda de agua de buena calidad, poniendo mayor énfasis en las medidas de control de la contaminación.

La introducción de tarifas o precios que reflejen el verdadero coste del recurso es una de las principales innovaciones de la DMA. Así, esta directiva propone disminuir el consumo de agua de riego mediante la aplicación de la tarifación como principal instrumento económico incentivador de su uso racional, evitando el deterioro cuantitativo y cualitativo de las masas de agua. Así pues, la tarifación se constituye en el seno de UE como el principal instrumento de las políticas de demanda antes comentadas, el cual será reforzado con otras medidas igualmente encaminadas a asegurar la asignación eficiente del agua dentro del sector agrario, tales como la modernización del regadío o los mercados de agua (Dinar, 2000).

Dada la notoriedad alcanzada por la tarifación del agua tras la aprobación de la DMA, la comunidad científica ha abordado el tema extensamente, tratando de analizar por anticipado los previsibles impactos (análisis ex-ante) de la implementación de este instrumento económico en la agricultura de regadío. En la mayoría de las ocasiones estos trabajos empíricos se han basado en modelos de programación matemática (PM), como la técnica que permite una mejor aproximación a la simulación de políticas en el ámbito agrario. Así, en el ámbito internacional podemos encontrar trabajos como los de Wheeler *et al*. (2008), Medellín-Azuara *et al*. (2012) o Grafton *et al*. (2015). De especial interés es el trabajo de Dinar *et al*. (2015), donde recogen las experiencias de tarifación empleadas en varios países del mundo, documentando los últimos 10-15 años de experiencias de fijación de precios e innovaciones en tarifación tales como la aplicación de reformas políticas, la recuperación de costes y la eficiencia y equidad en el uso del agua.

En el ámbito español los trabajos en esta línea han sido igualmente numerosos, destacando entre ellos los de Varela-Ortega *et al*. (1998), Berbel y Gómez-Limón

(2000), Mejías *et al*. (2004) e Iglesias y Blanco (2008). A pesar de ello, el análisis de los impactos de la tarifación requiere todavía de una mayor evidencia empírica, especialmente sobre sistemas de regadío con cultivos de alto valor añadido. Asimismo, debe indicarse que la literatura existente se basa en modelos que simulan los efectos de la tarifación bajo antiguos escenarios de la Política Agraria Común (PAC), muy diferentes del aplicable en la actualidad. Este motivo justifica igualmente la necesidad de nuevos trabajos empíricos en esta línea, que reporten evidencias actualizadas al respecto.

Dentro de este contexto, este trabajo trata de cubrir estas lagunas de conocimiento arriba comentadas, analizando el efecto que tendría la tarifación del agua sobre la agricultura de regadío en sistemas de cultivos más intensivos, tomando como caso de estudio la Comunidad de Regantes (CR) del Sector BXII en el Bajo Guadalquivir. Además, este trabajo aborda una simulación que tiene en cuenta las novedades introducidas por la reforma de la PAC que han entrado en vigor en el año 2015, al objeto de evidenciar si tal cambio normativo tiene influencia sobre los impactos previsibles de la tarifación del agua de riego. Con este objetivo se han construido tres modelos de simulación, basados en tres métodos de programación matemática, los cuales han sido aplicados a tres "explotaciones tipo" características de la zona de estudio. De esta manera se ha tratado de simular la toma de decisiones de los regantes acerca de la adopción de planes de cultivo y técnicas de riego frente a la tarifación, y así poder analizar los impactos socioeconómicos de este instrumento económico.

2.2. Material y métodos

2.2.1. Descripción de la zona

La CR del Sector BXII del Bajo Guadalquivir es una zona regable de 14.643 ha situada en la margen izquierda del río Guadalquivir, cerca de su desembocadura en el océano Atlántico, repartida entre los municipios sevillanos de Lebrija y Las Cabezas de San Juan. Se trata de una zona anteriormente de marismas, improductiva para agricultura, que fue desecada y transformada en regadío por el Instituto Nacional de Reforma y Desarrollo Agrario (IRYDA) durante la década de los setenta del siglo pasado. A medida que se fue terminando la transformación (entre 1980 y

1990), estas nuevas tierras fueron dividas en parcelas perfectamente rectangulares de 12 ha y entregadas a 1.142 colonos. Este proceso de creación ha permitido que las explotaciones que componen la zona regable presenten una relativa homogeneidad, ya que comparten clima, edafología, técnicas de cultivo y tienen un tamaño igualmente similar. Asimismo, el proceso de colonización seguido ha hecho que el perfil sociodemográfico de los titulares sea igualmente homogéneo, presentando perfiles similares en cuanto a edad, formación y renta.

La zona regable cuenta con una concesión otorgada por la Confederación Hidrográfica del Guadalquivir (CHG) de 6.700 m^3/ha anuales. El agua se extrae del propio cauce del río, siendo transportada hasta la zona regable a través del Canal del Bajo Guadalquivir. Una vez en la CR, el agua es distribuida entre las diferentes explotaciones mediante una red de tuberías presurizadas completamente modernizadas que permite un sistema de riego a la demanda. Los principales cultivos de la zona son algodón (44,5% del total de la superficie de la CR), maíz (12,9%), tomate (11,5%), remolacha (9,4%), trigo (8,7%), girasol (6,8%) y hortícolas como la zanahoria y la cebolla (6,3%). Estos cultivos son principalmente regados mediante aspersión (72% de la superficie de la CR), aunque también se emplean las técnicas de goteo (22%) y riego superficial (6%).

Actualmente la zona regable del Sector BXII está dividida en 569 explotaciones, con un tamaño medio de 25,7 ha. Los servicios prestados por la CR se tarifan de forma binómica. Durante la campaña 2013, el importe facturado a los comuneros consistió en una derrama por superficie de 294 €/ha (por el canon y la tarifa pagados a la CHG, los gastos de personal, los gastos generales y los gastos de mantenimiento), a la que se sumó un cargo en función del consumo de agua de 0,0023 €/m^3 (por el consumo de energía).

Los datos anteriores evidencian que se trata de una zona con alto grado de modernización tecnológica, así como como una producción altamente rentable. Como se ha comentado anteriormente, en la literatura apenas hay estudios sobre el impacto de la tarifación en zonas regables con estas características, circunstancia que justifica la elección de la CR del Sector BXII como zona de estudio para este trabajo empírico.

2.2.2. *Captura de datos*

El uso del agua en el regadío depende esencialmente de las decisiones que tomen los regantes, y más concretamente de las decisiones relativas al plan de cultivos (alternativas a sembrar y técnicas de riego a emplear). Esta circunstancia justifica que la unidad más adecuada para el análisis del impacto de la tarifación sea la explotación agraria, considerada como la unidad de gestión de la agricultura de regadío, donde el uso de agua está condicionado por la toma de decisiones de su titular.

La caracterización de las explotaciones de regadío del Sector BXII se ha realizado mediante una encuesta realizada específicamente para este estudio, gracias a la cual se ha recabado información sobre los planes de cultivos, sistemas y dosis de riego, y otros aspectos relacionados con la gestión del agua. Para la realización de esta encuesta se ha extraído una muestra representativa de regantes de la zona de estudio, seleccionados mediante rutas aleatorias, respetando cuotas por estratos de tamaños. Aunque el tamaño muestral inicial se fijó en 60 individuos, tras las correspondientes entrevistas personales se ha podido contar con 59 cuestionarios válidos. El trabajo de campo correspondiente se realizó durante los meses de marzo y abril de 2014.

Asimismo, se ha realizado un estudio de la rentabilidad (ingresos y gastos) de los cultivos presentes en la zona regable. Para ello se ha partido de información tanto primaria como secundaria. Las fuentes primarias, basadas en entrevistas a técnicos y agricultores de la zona de estudio, han permitido configurar los itinerarios tecnológicos (prácticas de cultivo y dosis de insumos) de los diferentes cultivos, así como los costes unitarios de producción (precios pagados). Con toda esta información se ha podido calcular los ingresos, gastos e indicadores de rentabilidad de cada alternativa de cultivo para el año base (2013). Por otra parte, se han empleado datos secundarios (anuarios de estadística agraria del MAGRAMA y de la Junta de Andalucía, e información del Fondo Español de Garantía Agraria) para obtener las series históricas de rendimientos, precios y subvenciones, las cuales nos han permitido igualmente generar una serie histórica (2007-2013) de ingresos, gastos e indicadores de rentabilidad de los diferentes cultivos.

Los datos recogidos en la encuesta han puesto de manifiesto ciertas disparidades de planes de cultivos y técnicas de riego empleadas entre los regantes de la zona de estudio. Estas disparidades, común a la mayoría de sistemas agrarios de regadío, hace recomendable que el estudio empírico del uso del agua se base en los análisis individualizados de diferentes "explotaciones tipo", representativas de la diversidad de explotaciones existentes en la zona regable. Con el propósito de definir el número y características de las estas explotaciones tipo, en este trabajo se ha empleado la técnica del análisis de grupos, de conglomerados o clúster, tal y como recomiendan Gómez-Limón y Riesgo (2004). La técnica de conglomerados se utiliza para agrupar objetos o individuos que se consideran similares entre sí a partir de un conjunto de variables que los caracterizan. En nuestro caso, los elementos a tipificar son las 59 explotaciones de la muestra, empleando para ello el plan de cultivos (porcentaje de superficie dedicada a cada cultivo) y las técnicas de riego utilizadas como variables tipificadoras. Dentro de las posibilidades que permite esta técnica, se ha optado en este caso por emplear el método de Ward como criterio de agregación y la distancia euclídea como medida de la distancia entre elementos de la muestra. De esta manera se han obtenido finalmente tres clases o clústeres, donde se agrupan la totalidad de las explotaciones muestreadas. A partir de esta tipología se ha podido caracterizar las explotaciones "tipo" resultantes mediante los valores medios para cada clase de las diferentes variables recogidas en las encuestas, tanto las relativas al plan de cultivos (variables tipificadoras) como otras relativas a aspectos estructurales de las mismas (tamaño y características sociodemográficas del titular: género, edad, formación, etc.).

En la Tabla 2.1 se resume los resultados del análisis clúster realizado, donde se indica el plan de cultivos de cada explotación tipo resultante (variables empleadas para la tipificación), así como el tamaño de las mismas, única variable estructural donde se han encontrado diferencias significativas entre clústeres.

Tabla 2.1. *Resultados del análisis clúster: características de las explotaciones tipo*

	Explotación tipo 1	*Explotación tipo 2*	*Explotación tipo 3*
Variables tipificadoras (alternativa de cultivos-sistema de riego)			
Trigo blando aspersión (%)	5,4	6,4	3,9
Maíz superficie (%)	1,8	1,2	
Maíz aspersión (%)		3,8	
Maíz goteo (%)	1,5	4,3	
Remolacha aspersión (%)	24,0	6,2	39,1
Algodón superficie (%)		2,9	6,7
Algodón aspersión (%)	29,6	54,4	50,3
Girasol superficie (%)		2,1	
Alfalfa aspersión (%)	0,5	3,8	
Tomate goteo (%)	30,3	12,8	
Cebolla aspersión (%)	1,6	0,5	
Zanahoria aspersión (%)	5,3	2,0	
Variables estructurales con diferencias significativa			
Superficie total (ha)	35,8	23,9	15,0

Las características de cada una de las explotaciones tipo así definidas se resumen como sigue:

- *Explotación tipo 1*: “*Grandes regantes profesionales*”. Representa al 39% de explotaciones de la muestra, que ocupan el 52% de la superficie total de la zona regable. Se trata de la explotación tipo de mayor tamaño de la CR (35,8 ha), orientada principalmente hacia cultivos hortícolas (en torno al 40% de su superficie: tomate por goteo, 30,3%; zanahoria, 5,3%; y cebolla, 1,6% regados por aspersión), que son los cultivos más rentables. Otros cultivos de importancia en su plan de cultivos son algodón (29,6%) y remolacha azucarera (24,0%), ambos regados por aspersión.

- *Explotación tipo 2*: “*Diversificadores de riesgo*”. Esta explotación representa al 41% de los agricultores de la muestra, y ocupa el 36% de la superficie de la CR. Se trata de una explotación de tamaño mediano (23,9 ha), caracterizada por tener un plan de cultivos altamente diversificado, dedicando superficies a todas las alternativas presentes en la zona de estudio y utilizando todas las técnicas de riego: superficie, aspersión y goteo.

- *Explotación tipo 3*: “*Agricultores conservadores extensivos*”. Este tipo de explotación representa la menor proporción de los agricultores de la muestra

(20%), y cubre solo el 11% de la superficie total de la CR. Se trata pues de la explotación tipo de menor tamaño, con una media de 15 ha. Su superficie agraria está ocupada por los cultivos más extensivos de la zona, como algodón (57,0%), regado principalmente por aspersión, aunque también es regado por superficie, y remolacha por aspersión (39,1%), ambos muy dependientes de los subsidios de la PAC.

2.2.3. Modelización a través de la programación matemática

La cuantificación de los impactos socioeconómicos de la tarifación del agua de riego requiere de la consideración de diferentes escenarios de precios del recurso, al objeto de poder simular el comportamiento (toma de decisiones) de los regantes frente a los mismos. En este sentido, el enfoque metodológico más adecuado son los modelos de programación matemática, ya que permite incluir una gran cantidad de información económica y técnica con un nivel de desagregación apropiado, para poder representar y analizar diferentes escenarios (Hazell y Norton, 1986).

En este trabajo la aplicación de la programación matemática (PM) ha consistido en la construcción de modelos a través de los cuales simular individualmente el comportamiento de cada una de las tres explotaciones tipo definidas frente a tarifas volumétricas crecientes del agua de riego. De manera más concreta, cabe comentar que se han construido un total de 3 modelos para cada una de las explotaciones tipo, utilizando para ello tres técnicas de PM diferentes: a) maximización del beneficio, b) Programación Matemática Positiva (PMP), y c) maximización de una Función de Utilidad Multiatributo (MAUF). Así pues, se han construido un total de 9 modelos (3 explotaciones tipo × 3 métodos de PM). La consideración de tres métodos diferentes de PM para la realización del estudio empírico está justificada por el propósito de obtener resultados y conclusiones más robustos.

Una vez que los modelos se han construido, estos se han adaptado para la simulación del comportamiento de las explotaciones tipo frente la tarifación del agua de riego. Para ello se ha incluido en dichos modelos la tarifa del agua (P_w) como un nuevo parámetro, que se ha hecho variar en un intervalo de 0,00 €/m³ a 1,00 €/m³. Así, considerando la tarifa volumétrica del agua como un nuevo coste variable de producción (dependiente de las necesidades de agua de los cultivos sembrados), se ha derivado el plan de cultivos óptimo para cada uno de los 101 escenarios de precios del agua considerados (parametrización de P_w con incrementos sucesivos de 0,01€/m³). A partir de estos resultados de los modelos de PM se han obtenido, igualmente para cada nivel de tarifación, el valor de determinadas variables de interés para el análisis político de este instrumento económico: a) la cantidad de agua demandada por el regante, b) el beneficio de los agricultores, c) el empleo generado en la agricultura, d) la recaudación pública proveniente de la tarifa del agua, y e) el nivel de bienestar económico generado por el uso agrario del agua.

El último paso realizado ha consistido en agregar los resultados obtenidos para las explotaciones tipo, al objeto de estimar así los resultados agregados a nivel del conjunto de la zona regable. Con este propósito se ha realizado una suma ponderada de las variables obtenidas en cada explotación tipo, considerando para ello la superficie representada por cada una de ellas.

Los modelos de PM tienen tres elementos definitorios: a) variables de decisión, b) restricciones, y c) función objetivo. A continuación, se detalla cómo se ha considerado cada uno de estos elementos en la construcción de los modelos empleados en este trabajo.

a) **Variables de decisión**. Se han considerado como variables de decisión la superficie dedicada a cada una de las alternativas de cultivo-sistema de riego existentes en la zona, $\vec{x}_{i,j} = (x_{1,1}, \dots x_{i,j}, \dots, x_{n,m})$, donde $x_{i,j}$ es área dedicada a cada combinación de cultivo i y técnica de riego j. En la primera columna de Tabla 2.1 pueden observarse las alternativas de cultivos empleadas como variables de decisión en los modelos, construidos como binomios de los cultivos y las técnicas de riego conforme a las existentes en la zona de estudio. Además, como variables de decisión posibles en un escenario de tarifación del agua de riego, se ha considerado igualmente las alternativas de cultivos de secano (trigo y girasol), dejar la tierra en barbecho y el arrendamiento de tierra de pastizal a terceros como posibilidad para cumplir con las exigencias de la nueva PAC.

En este sentido conviene indicar que el riego deficitario no ha sido considerado a la hora de definir las variables de decisión, debido a que en la zona de estudio este tipo de manejo del riego no es considerado en la práctica por los agricultores, dado el carácter anual de los cultivos existente en la misma. Efectivamente, en el caso de los cultivos herbáceos existe una relación lineal entre la evapotranspiración y el rendimiento del cultivo, circunstancia que provoca que, desde una perspectiva técnica y económica, la dosis óptima de agua de riego coincida con el máximo técnico (necesidades hídricas de los cultivos completas), con independencia de la tarifa de agua aplicada, siempre y cuando el cultivo genere una rentabilidad positiva. Este hecho justifica la nula incidencia del riego deficitario en cultivos herbáceos, tal y como ha quedado demostrada en diversos trabajos empíricos (véase, por ejemplo, Martínez-Álvarez *et al*., 2014).

b) **Restricciones**. Son las condiciones o los límites, tanto superiores como inferiores, que deben respetar las variables de decisión, ya que de lo contrario la solución sería imposible de implementar en el mundo real. Para todos los modelos desarrollados se ha considerado las siguientes restricciones:

- Restricciones de *ocupación de la tierra*. La suma de todas las variables de decisión tiene que ser menor o igual que la superficie disponible en la explotación tipo k (a_k), de tal manera que se verifique que:

$$\sum_{i=1}^{n}\sum_{j=1}^{m}\vec{x}_{i,j} \leq a_k \tag{1}$$

- Restricciones de *agua*. Los requerimientos de agua de los cultivos no pueden superar la cantidad de agua disponible por explotación, que se deriva de la superficie de la misma (a_k) y de la concesión otorgada a la CR por la CHG (b_k= 6.700 m^3/ha). Así, se tiene que cumplir que:

$$\sum_{i=1}^{n}\sum_{j=1}^{m}\vec{x}_{i,j} \cdot wr_{i,j} \leq a_k \cdot b_k \tag{2}$$

  donde $wr_{i,j}$ son los requerimientos de agua de la alternativa i y técnica de riego j.

- Restricciones de *mercado*. Algunos cultivos, por su carácter perecedero (principalmente los hortícolas), presentan limitaciones en su comercialización, dado que el mercado no puede absorber una cantidad superior a la demanda. Por este motivo se han incluido restricciones que limitan las superficies cultivadas de tomate, cebolla y zanahoria a la máxima superficie histórica de cada explotación tipo k durante el período 2007-2013.

- *Cupo remolacha*. Según el sistema de cuotas de producción establecido por la PAC, la superficie cultivada con remolacha azucarera no puede superar la superficie observada, que es equivalente a la cuota asignada a cada explotación tipo.
- *Restricciones agronómicas*. Se han incluido las restricciones de sucesión y rotación de cultivos consideradas de la zona de estudio, de acuerdo con la información proporcionada por los técnicos entrevistados.
- Restricciones relativas al *sistema de riego*. Se supone que las infraestructuras específicas de cada sistema de riego (superficie, aspersión y goteo) se mantienen fijas en el corto plazo. Como consecuencia, se ha asumido que el área regada por cada uno de estos sistemas en cada tipo de explotación no se puede variar en más de 10% respecto la superficie observada.

$$\sum_{i=1}^{n} \vec{x}_{i,j} \leq 1{,}1 * \sum_{i=1}^{n} \vec{x}_{i,j}^{obs} \qquad \forall j \tag{3}$$

La aplicación de la PAC establece igualmente una limitación legal en relación con la superficie máxima individual de algodón. No obstante, tras la información recogida durante el trabajo de campo, se ha considerado pertinente no incluir esta limitación como restricción de los modelos, ya que los titulares de explotación de la zona analizada han venido sorteando dicha limitación para poder acogerse a la ayuda del cultivo del algodón, compartiendo la titularidad de las explotaciones con otros miembros de su familia.

Tal y como se han definido las variables de decisión y las restricciones para todos los modelos, resulta evidente que estos se han planteado para la realización de simulaciones considerando un horizonte temporal a corto plazo. Este enfoque cortoplacista está justificado por el objetivo definido para el trabajo, consistente en analizar el efecto que supondría incrementos en la tarifa del agua dentro del contexto de la nueva PAC. Esto implica que las únicas variables sobre la que puede decidir el agricultor son las relativas al plan de cultivos (se considera que los bienes de capital fijo –p. ej., los sistemas de riego– y la dimensión de las explotaciones son componentes estructurales que no pueden alterarse).

c) **Función objetivo**. Es la expresión matemática que el agricultor trata de optimizar a través de su toma de decisiones (asignación de valores a las variables de decisión). Dicha expresión se fija en función de los supuestos de partida que se asuman sobre el comportamiento de estos productores. En este trabajo se ha optado por considerar diferentes supuestos al respecto, derivándose por tanto diferentes expresiones para la función objetivo. Así, la expresión de dicha función es el elemento diferenciador de los tres métodos de PM utilizados. A continuación, se describen las funciones objetivo empleadas para cada uno de los tres métodos aplicados.

2.2.4. Maximización del beneficio

En este caso, siguiendo el criterio clásico de la Economía de la empresa, la función objetivo considerada es la maximización del beneficio. Dado el ámbito decisional de los modelos de simulación desarrollados, que pretenden simular decisiones de los regantes a corto plazo (es decir, las relativas al plan de cultivo), se ha optado por considerar el margen bruto total de la explotación (MBT) como como proxy operativo para cuantificar las variaciones en el beneficio, variaciones que a la postre son las que determinan los cambios en las decisiones productivas. Este indicador económico se calcula como diferencia de los ingresos de explotación (ventas y subvenciones acopladas) y los costes variables.

Matemáticamente, el margen bruto de cada alternativa de cultivo i y técnica de riego j $MB_{i,j}$ se calcula como sigue:

$$MB_{i,j} = \sum_{i=1}^{n} \sum\nolimits_{j=1}^{m} \left[Y_{i,j} \cdot P_i + S_i - CV_{i,j} - (wr_{i,j} \cdot P_w) \right] \quad (4)$$

Siendo:

$Y_{i,j}$ = rendimiento del cultivo i y técnica de riego j (en kg/ha)
P_i = precio de venta del cultivo i (en €/kg)
S_i = ayuda acoplada a la producción del cultivo i (en €/ha)

$CV_{i,j}$ = costes de producción del cultivo i y técnica de riego j (en €/ha)
$wr_{i,j}$ = requerimiento de agua del cultivo i y técnica de riego j (en m^3/ha)
P_w = tarifa del agua (en €/m^3)

La función objetivo a maximizar en este caso será, por tanto, la siguiente:

$$MBT = \sum_{i=1}^{n} \sum_{j=1}^{m} MB_{i,j} \cdot \vec{x}_{i,j} \tag{5}$$

2.2.5. *Programación Matemática Positiva (PMP)*

La PMP, desarrollada por Howitt (1995), también asume un comportamiento maximizador del beneficio de los agricultores, pero, a diferencia del enfoque clásico, considera una función objetivo no lineal, obtenida mediante un proceso de calibración que ajusta la función de costes de los cultivos mediante una función cuadrática. Esta innovación metodológica permite simulaciones más flexibles y realistas que con la maximización del beneficio. Este hecho le ha convertido en uno de los enfoques de PM más aceptados y utilizados para la modelización del sector agrario.

Para la aplicación de la PMP a nuestro caso de estudio se ha optado por seguir el enfoque estándar propuesto por Howitt (1995). No obstante, dentro de este enfoque estándar se ha considerado la aproximación del coste medio (average cost approach) propuesta por Heckelei y Britz (2000), que considera que los costes medios de la función de costes variables para cada actividad son iguales a los costes observados. De esta manera el margen bruto tras aplicar la PMP resulta ser igual al margen bruto observado.

La aplicación de este enfoque estándar de la PMP consta de dos fases. La primera de ellas consiste en la construcción de un modelo de maximización del beneficio como el antes comentado, al cual se le añaden una serie de restricciones adicionales (restricciones de calibración), que limitan la superficie destinada a cada cultivo a las superficies observadas en la realidad. Al incluir tales restricciones se fuerza a que la solución óptima del modelo reproduzca exactamente los niveles de actividad (superficie de cultivo y técnica de riego) observados en el año base. En la segunda fase, los valores duales de las restricciones de calibración se utilizan para especificar una función objetivo de maximización de beneficio con costes cuadráticos, tal y como se describe a continuación:

$$Max\ MB = \sum_{i=1}^{n}\sum_{j=1}^{m}(P_i \cdot \vec{x}_{i,j} + S_i - \alpha_{i,j} \cdot \vec{x}_{i,j} - \frac{1}{2}\ \beta_{i,j} \cdot \vec{x}_{i,j}^{2}) \tag{6.1}$$

$$\alpha_{i,j} = CV_{i,j} - \lambda_{i,j} \tag{6.2}$$

$$\beta_{i,j} = \frac{2\lambda_{i,j}}{x_{i,j}^{obs}} \tag{6.2}$$

donde $\lambda_{i,j}$ es el precio sombra (valor dual) de la restricción de calibración del cultivo i y técnica de riego j, y $\alpha_{i,j}$ y $\beta_{i,j}$ son los parámetros de calibración de la PMP.

La función objetivo así generada incluye una función de costes cuadrática que trata de recoger todos los costes de los diferentes cultivos, tanto los observables como los no observables. El interés de esta función objetivo es que su maximización permite reproducir de manera exacta el plan de cultivo de las explotaciones modelizadas para el escenario base empleado para la calibración. Esta circunstancia suele esgrimirse como argumento en favor de este enfoque de PM, ya que permite asumir que las simulaciones realizadas mediante esta técnica son más realistas que las derivadas de otros enfoques que no son capaces de reproducir ni tan siquiera las soluciones del escenario base.

Cabe comentar que la PMP ha evolucionado considerablemente en los últimos años, planteándose nuevos desarrollos que tratan resolver algunas de las debilidades del enfoque estándar propuesto por Howitt (1995). Entre estos desarrollos cabe comentar el de Röhm y Dabbert (2003), que considera un intercambio más favorable entre "variantes" de determinadas actividades (p. ej., distintos sistemas de riego para un mismo cultivo en nuestro caso) que entre actividades diferentes, o el de Cortignani y Severini (2009), que posibilita la calibración de alternativas no observadas en el año base, (p. ej., las alternativas de secano en nuestro caso). Todos estos desarrollos se basan en procedimientos de calibración que utilizan un valor marginal de la tierra en el escenario base. Por este motivo el empleo de estas variantes de la PMP para la simulación de escenarios es sólo recomendable en aquellos casos en que el valor marginal de la tierra (renta de la tierra) no se vea afectado de manera significativa (no tiene sentido que funciones objetivo calibradas en base a la renta de la tierra del escenario base se utilicen para simular escenarios donde dicha renta es radicalmente diferente). Desgraciadamente, este no es el caso de la tarifación del agua de riego, instrumento que supone la adición de un coste lineal a cada uno de los cultivos regadío, y que por tanto provoca la disminución de la renta de la tierra. Por este motivo, se ha tenido que descartar el uso de los desarrollos de la PMP arriba comentados para esta investigación, optándose por el enfoque estándar de la PMP como mejor opción disponible.

2.2.6. Maximización de una Función de Utilidad Multiatributo (MAUF)

Este método de PM asume que el comportamiento de los agricultores trata de maximizar una *Función de Utilidad Multiatributo* (MAUF en inglés) en la que condensan todos los criterios que estos productores consideran relevantes (Romero y Rehman, 2003). Este enfoque multicriterio se supone igualmente más realista que la simple maximización del beneficio, pues se trata de un modelo más general.

Para la aplicación de este enfoque a nuestro caso de estudio se asume que los principales criterios que integran la función de utilidad de los agricultores (Gómez-Limón y Berbel, 2000; Gómez-Limón *et al.*, 2004; Gutiérrez-Martín y Gómez-Gómez, 2011) son:

- La *maximización del beneficio*:

$$MBT = \sum_{i=1}^{n} \sum_{j=1}^{m} MB_{i,j} \cdot \vec{x}_{i,j} \qquad (7)$$

- La *minimización del riesgo* medido como la varianza del margen bruto (VAR) durante un período de 7 años (2007-2013). Así, el riesgo se calcula como:

$$VAR = \vec{x}_{i,j}^{t}\,[cov]\,\vec{x}_{i,j} \qquad (8)$$

  donde $[cov]$ es la matriz de varianzas-covarianzas de los márgenes brutos de los cultivos por hectárea en el período considerado.

- La *minimización de la mano de obra total* (MOT) empleada en la explotación, considerada como proxy de la complejidad de gestión. Matemáticamente este atributo se calcula como:

$$MOT = \sum_{i=1}^{n} \sum_{j=1}^{m} MO_{i,j} \cdot \vec{x}_{i,j} \tag{9}$$

donde $MO_{i,j}$ es la cantidad de mano de obra demanda de cada alternativa i, j.

La formulación matemática concreta de la función de utilidad se ha obtenido siguiendo la metodología propuesta por Sumpsi *et al*. (1996) basada en una función de utilidad aditiva:

$$MAUF = \sum_{a=1}^{l} w_a \, u_a(f_a) \tag{10}$$

donde $u_a(f_a)$ es la expresión matemática de la utilidad parcial asociada al atributo a, y w_a es la ponderación o peso otorgado dicho atributo a, cumpliéndose que:

$$\sum_{a=1}^{l} w_a = 1 \tag{11}$$

Para la determinación operativa de la MAUF característica del comportamiento de los regantes, se han considerado funciones de utilidad parcial monoatributo ($u_a(f_a)$) lineales, normalizadas en todos los casos para tomar valores entre 0 (menor utilidad posible) y 1 (mayor utilidad posible). Para ello se empleado la expresión $u_a = \frac{f_a(\vec{x}_{i,j}) - f_{a*}}{f_a{}^* - f_{a_*}}$ para los atributos del tipo "más es mejor", y la expresión $u_a = \frac{f_{a*} - f_a(\vec{x}_{i,j})}{f_{a_*} - f_a{}^*}$ para los atributos de tipo "menos es mejor", donde $f_a{}^*$ y f_{a_*} son los valores ideales y anti-ideales, respectivamente, de cada atributo en la correspondiente matriz de pagos. Así las funciones monoatributo de los criterios asociadas a MBT, VAR y MOT han quedado como sigue:

$$u_{MBT} = MBTn = \frac{MBT(\vec{x}_{i,j}) - MBT_*}{MBT^* - MBT_*} \tag{12}$$

$$u_{VAR} = VARn = \frac{VAR_* - VAR(\vec{x}_{i,j})}{VAR_* - VAR^*} \tag{13}$$

$$u_{MOT} = MOTn = \frac{MOT_* - MOT(\vec{x}_{i,j})}{MOT_* - MOT^*} \tag{14}$$

Puede evidenciarse, que todas las $u_a(f_a)$ así obtenidas son del tipo "más es mejor", por lo que $VARn$ debe interpretarse como el riesgo evitado y $MOTn$ como la complejidad de gestión evitada.

Siguiendo el método propuesto por Sumpsi *et al*. (1996), el valor de las ponderaciones w_a se obtiene a partir de un modelo auxiliar de programación por metas ponderadas, que trata de reproducir con exactitud el comportamiento observado de los productores. Así, finalmente se obtiene una función de utilidad del tipo:

$$MAUF = w_1 \cdot MBTn + w_2 \cdot VARn + w_3 \cdot MOTn \qquad (15)$$

Debe indicarse en cualquier caso, que la forma funcional de MAUF será diferente para cada explotación tipo. Efectivamente, este enfoque de PM permite detectar diferencias en las ponderaciones de los diferentes atributos en base a la disparidad de los planes de cultivos observados en el escenario base. Esta circunstancia se ha esgrimido igualmente como una ventaja singular del mismo, que justifica la obtención de soluciones más realistas.

2.2.7. Adaptación de los modelos a la PAC-2015

Las observaciones de las decisiones de los regantes en el año base se corresponden con el año 2013, decisiones que estuvieron condicionadas por la PAC aplicada en ese año (PAC-2013). Sin embargo, debe tenerse en cuenta que la simulación del comportamiento futuro de los regantes requiere la consideración del nuevo marco normativo de la PAC, resultante de la última reforma de esta política europea y que ha entrado en vigor en 2015 (PAC-2015).

Esta circunstancia ha obligado a considerar de dos escenarios distintos de política agraria. Así, las calibraciones de las funciones objetivo de los modelos PMP y MAUF se han realizado considerando el escenario PAC-2013, teniendo en cuenta las subvenciones acopladas (S_i) existentes entonces, así como las restricciones arriba indicadas.

Sin embargo, para simular el comportamiento de los regantes frente a la tarifación del agua de riego ha exigido la consideración del escenario actual de la PAC (PAC-2015). Para ello se han tenido que adaptar los modelos empleados para la calibración, al objeto de actualizar el nivel de subvenciones acopladas y tener en cuenta las novedades en relación al nuevo "pago verde", tal y como se describe a continuación:

- Las explotaciones mayores de 15 ha deben dedicar al menos un 5% de la explotación a superficies de interés ecológico o SIE (barbecho, cultivos fijadores de nitrógeno y forestadas). Así, si $a_k \geq 15$, entonces debe cumplirse que $SIE \geq 0{,}05 \cdot a_k$.
- Las explotaciones entre 10 y 30 ha deben tener al menos dos cultivos diferentes, siendo el cultivo principal menor del 75% de la superficie de la explotación. Así, si $10 \leq a_k \leq 30$, entonces debe verificarse que $\sum_{j=1}^{m} \cdot \vec{x}_{i,j} \leq 0{,}75 \cdot a_k$ para todo i.
- Las explotaciones mayores de 30 ha tienen que tener al menos tres cultivos diferentes, donde el cultivo principal debe ocupar menos del 75%, y la suma de los dos cultivos principales tiene que ser menor del 95%. Así, si $a_k > 30$, entonces ha de cumplirse, además de $\sum_{j=1}^{m} \cdot \vec{x}_{i,j} \leq 0{,}75 \cdot a_k$ para todo i, y que $\sum_{j=1}^{m} \cdot \vec{x}_{i,j} + \sum_{j=1}^{m} \cdot \vec{x}_{i',j} \leq 0{,}95 \cdot a_k$ para todo $i \neq i'$.

2.3. Resultados

En primer lugar, se han obtenido las funciones de utilidad para cada una de las explotaciones tipo consideradas por el método MAUF, resultado las funciones que se muestran a continuación:

Explotación tipo 1: $MAUF_1 = 0{,}766 \cdot MBTn + 0{,}234 \cdot VARn$ (16)

Explotación tipo 2: $MAUF_2 = 0{,}775 \cdot MBTn + 0{,}225 \cdot VARn$ (17)

Explotación tipo 3: $MAUF_3 = 0{,}848 \cdot MBTn + 0{,}122 \cdot VARn + 0{,}030 \cdot MOTn$ (18)

Observando las diferentes MAUF se evidencia como el margen bruto tiene en todos los casos una importancia relativa muy superior al resto de los criterios. La similitud de comportamiento (MAUF) puede explicarse por la relativa homogeneidad de las explotaciones (clima, edafología, tamaño y técnicas de cultivo) y titulares (edad, formación, etc.) de la zona regable analizada, tal y como se comentaba en la sección dedicada a la descripción del caso de estudio.

Los modelos construidos para cada explotación tipo han permitido obtener, para cada nivel de la tarifa volumétrica del agua (parametrización de P_w desde 0,00 a 1,00 €/m³), los correspondientes planes de cultivos óptimos como primeros resultados. A partir de esta información sobre el previsible comportamiento de los regantes ante cada escenario de precios del agua, se han podido obtener una serie de variables de interés para la toma de decisión política: el margen bruto total generado por la actividad agraria, la recaudación pública derivada de la tarifación y el nivel de bienestar económico derivado del uso del agua en la agricultura (indicadores económicos), el empleo generado por el regadío (indicador social) y el uso del agua para riego (indicador ambiental). El análisis de la evolución de estos indicadores a medida que se incrementa la tarifa del agua, sin duda, resulta de utilidad para la evaluación ex-ante de este instrumento económico, de cara a su adecuado diseño y gestión.

2.3.1. El efecto aislado de la reforma de la PAC

El primer resultado a reportar es el del previsible impacto de la reforma de la PAC-2015. Para ello se comparan el valor de los indicadores de interés político antes comentados en el escenario base (año 2013) con los simulados para un escenario PAC-2015 con precio del agua nulo, tal y como se muestra en la Tabla 2.2.

En esta tabla puede observarse cómo la última reforma de la PAC apenas va a afectar a la agricultura de la zona regable analizada. De hecho, las variaciones esperadas en el margen bruto, la demanda de mano de obra o el uso de agua de las diferentes explotaciones tipo son inferiores al 1% respecto a la situación actual. Estos resultados se explican por dos motivos. Primero, porque las condiciones de rotación obligatoria establecidas por el *greening* ya se cumplían previamente a la reforma y, por tanto, no ha sido necesario hacer cambios en los planes de cultivo por este motivo. Segundo, porque la estrategia a seguir por las explotaciones de la zona analizadas para el cumplimiento de la superficie de interés ecológico (SIE), que implica dedicar al menos un 5% de sus tierras a barbecho, cultivos fijadores de nitrógeno (leguminosas como la alfalfa), pastos o a superficie forestada, va a consistir en arrendar tierra de secano fuera de la zona regable para dedicarlas a barbecho o a pastos.

Estos resultados evidencian por tanto que la última reforma de la PAC no condiciona el potencial del impacto de la tarifación del agua de riego.

Tabla 2.2. *Impacto previsible de la reforma de la PAC-2015*

	Explotación tipo 1	*Explotación tipo 2*	*Explotación tipo 3*	*Zona regable*
PAC-2013				
Margen bruto (€/ha)	3.140	2.269	1.820	2.678
Mano de obra (h/ha)	34,07	20,95	27,09	28,64
Uso de agua (m³/ha)	5.774	5.176	5.502	5.531
Nº cultivos en la rotación	7	9	4	9
Superficie 1er cultivo (%)	35,8	32,6	41,1	28,5
Superficie 2º cultivo (%)	29,3	28,2	39	24,4
Alfalfa regadío (%)	0,6	4,5	0,0	1,9
SIE[1] en superficie de secano arrendada (%)	0,0	0,0	0,0	0,0
SIE total (%)	0,6	4,5	0,0	1,9
PAC-2015				
Margen bruto (€/ha)	3.129	2.267	1.807	2.670
Mano de obra (h/ha)	34,12	20,96	27,14	28,67
Uso de agua (m³/ha)	5.773	5.176	5.502	5.531
Nº cultivos en la rotación	8	10	5	10
Superficie 1er cultivo (%)	35,8	32,6	41,1	28,5
Superficie 2º cultivo (%)	29,3	28,2	39,0	24,4
Alfalfa regadío (%)	0,6	4,5	0,0	1,9
SIE[1] en superficie de secano arrendada (%)	4,4	0,5	5,0	3,1
SIE total (%)	5,0	5,0	5,0	5,0
Impacto PAC				
Margen bruto (€/ha)	-0,4%	-0,1%	-0,7%	-0,3%
Mano de obra (h/ha)	+0,1%	0,0%	+0,2%	+0,1%
Uso de agua (m³/ha)	0,0%	0,0%	0,0%	0,0%
Nº cultivos en la rotación	+14,3%	+11,1%	+25,0%	+11,1%
Superficie 1er cultivo (%)	0,0%	0,0%	0,0%	0,0%
Superficie 2º cultivo (%)	0,0%	0,0%	0,0%	0,0%
Alfalfa regadío (%)	0,0%	0,0%	0,0%	0,0%
SIE[1] en superficie de secano arrendada (%)	+4,4%	+0,5%	+5,0%	+3,1%
SIE total (%)	+4,4%	+0,5%	+5,0%	+3,1%

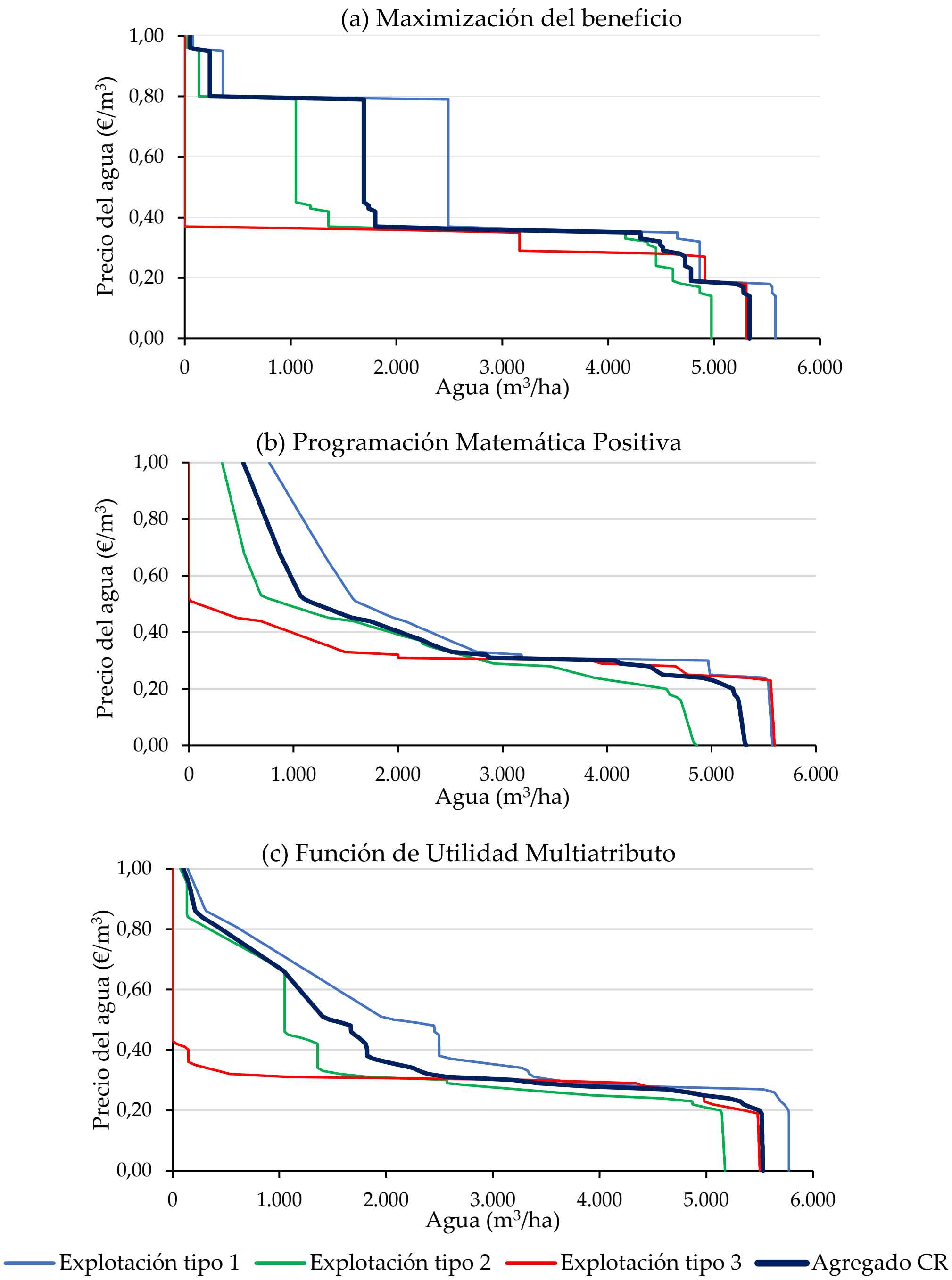
[1]Superfieicie de interés ecológico

2.3.2. *Las curvas de demanda de agua*

La presentación de resultados comienza con el análisis de la evolución del uso del agua para valores crecientes de la tarifa volumétrica de agua, a través de las correspondientes curvas de demanda, tal y como se evidencia en la Figura 2.1. En dicha figura pueden observarse ciertas diferencias en función tanto del método de PM empleado como de la explotación tipo analizada. En cuanto al método, se evidencia cómo las curvas derivadas a partir de la PMP y la MAUF experimentan cambios de pendiente más suaves, frente a la brusquedad en los cambios de la curva derivada de la maximización del beneficio. Esta circunstancia se debe a la mayor sensibilidad de la PMP y la MAUF a la hora simular escenarios, circunstancia que permiten considerar estos resultados más próximos al comportamiento real de los agricultores.

En relación con las explotaciones tipo, se observa cómo para tarifas bajas (entre 0,0 y 0,2 €/m^3) la explotación representativa del clúster 1 es en todos los casos la que presenta una mayor demanda de agua, seguida por la explotación tipo 3 y, por último, por la explotación tipo 2. Sin embargo, el comportamiento de cada una de ellas difiere a medida que se incrementa la tarifa de agua. Así, se observa como los "*Agricultores conservadores extensivos*" (explotación tipo 3) son los más reactivos a la tarifación, en la medida que son los que más reducen su consumo de agua cuando su precio sube, haciendo que para tarifas superiores a 0,3 €/m^3 su consumo sea prácticamente nulo. Por el contrario, los "*Grandes regantes profesionales*" (explotación tipo 1) se muestran más reticentes a reducir su consumo, demandando cantidades significativas de agua hasta niveles de la tarifa de 0,9 €/m^3. La diferencia en este comportamiento puede justificarse en las diferencias existentes en la productividad del agua en los cultivos incluidos en sus respectivos planes. Así, mientras en la explotación tipo 3 predominan cultivos extensivos con bajo valor añadido (p. ej., remolacha, algodón o maíz), con una baja productividad del uso del agua (productividad media aparente de 0,33 €/m^3), la explotación tipo 1 tiene una orientación principalmente hortícola, cultivos de mayor valor añadido y elevada productividad del agua (productividad media aparente de 0,53 €/m^3).

Figura 2.1. *Curvas de demanda de agua correspondientes a los tres métodos de programación matemática*



Asimismo conviene indicar que, a pesar de las diferencias arriba señaladas, para todos los métodos de PM empleados, y para todas las explotaciones tipo, se ha identificado un patrón similar en las correspondientes curvas de demanda, diferenciándose claramente dos tramos:

- *Tramo inelástico* (para tarifas de 0,0 a 0,3 €/m^3). La reacción de los regantes ante incrementos en el precio del agua no se corresponde con disminuciones en la demanda. Este comportamiento se debe a que para estas tarifas, incluso los cultivos con menor productividad del agua (remolacha, algodón y maíz) proporcionan una rentabilidad mayor que las alternativas de secano, circunstancia que justifica que los agricultores se resistan a hacer cambios en sus planes de cultivos (sustitución de cultivos con elevadas necesidades hídricas por otros que tengan un menor consumo de agua).
- *Tramo elástico* (para tarifas superiores a 0,3 €/m^3). Para precios superiores a 0,3 €/m^3 la curva de demanda adquiere una forma más elástica, es decir, ante subidas adicionales de la tarifa, los agricultores cambian progresiva y rápidamente sus planes de cultivo hacia alternativas con menor consumo hídrico, disminuyendo así el consumo total de agua y el coste correspondiente al uso del recurso.

2.3.3. Análisis de impactos socioeconómicos

Siguiendo con el análisis de los resultados, en la Figura 2.2 se puede observar la evolución de los indicadores socioeconómicos seleccionados para el estudio (margen bruto, empleo generado, recaudación por tarifación y eficiencia económica asociada al uso del agua en el regadío) a medida que se incrementa la tarifa del agua. A este respecto conviene indicar que, como ocurría con la demanda de agua, los resultados obtenidos a partir de los tres métodos de PM empleados siguen el mismo patrón de comportamiento. Por este motivo en la Figura 2.2 se ha optado, en aras de la simplicidad, por representar únicamente los resultados derivados del método MAUF.

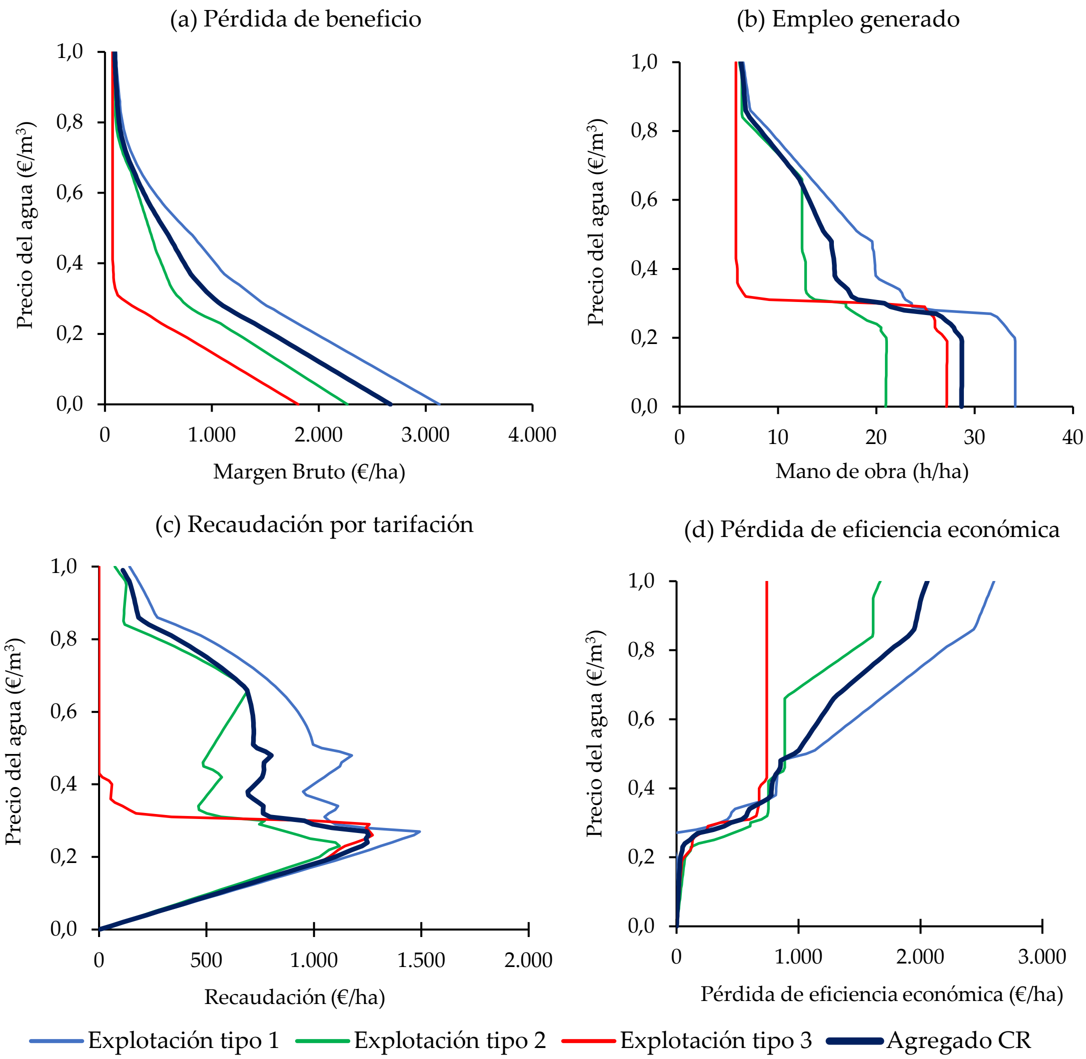
En líneas generales puede apreciarse como la tarifación progresiva del agua de riego supone una creciente pérdida de beneficios (medido por el margen bruto) por parte de los regantes, así como del empleo generado en el regadío (ver figuras 2.2a y

2.2b, respectivamente). Asimismo, se observa que la recaudación pública por la tarifación se incrementa a medida que aumenta la tarifa, hasta llegar a un nivel máximo que oscila entre 0,2 y 0,3 €/m^3 (ver Figura 2.2c), según los casos. Este punto de inflexión coincide con el cambio del tramo inelástico por el elástico antes descrito. Para tarifas superiores a este nivel, incrementos en el precio del agua se traducen en reducciones de la recaudación pública. Este hecho es importante destacarlo porque refleja que a partir de estos niveles de la tarifa del agua los regantes sustituyen rápidamente los cultivos de regadío por cultivos de secano, reduciendo de forma drástica la factura de agua como estrategia para afrontar el encarecimiento del uso del recurso.

En este sentido, puede observarse cómo durante el tramo inelástico (tarifas de 0,0 a 0,3 €/m^3), al no haber cambios en los planes de cultivos (no hay disminución en el uso del agua ni en el empleo), la pérdida de rentabilidad (margen bruto) por parte de los regantes equivale de manera exacta al incremento en la recaudación pública. Así, para estos niveles de precios del agua, la tarifación produce simplemente una transferencia de rentas desde el sector privado (productores de regadío) al sector público, sin que puedan significarse variaciones en el bienestar asociado al uso del agua de riego, cuantificado como suma del beneficio privado (margen bruto de las explotaciones) y público (recaudación). De esta manera, las pérdidas de eficiencia económica, considerando conjuntamente la parte privada como la pública, son prácticamente nulas (ver Figura 2.2d).

Una vez alcanzado el tramo elástico (tarifas que provocan cambios en los planes de cultivo) se puede advertir cómo tanto el consumo de agua como el margen bruto de las explotaciones y la generación de empleo disminuyen de manera importante. Por su parte, como ya se ha comentado, la recaudación pública disminuye por la drástica disminución del uso del agua. En consecuencia, puede afirmarse que para tarifas del agua superiores a 0,3 €/m^3 se aprecia una pérdida creciente de bienestar social, en la medida que la tarifación supone una pérdida tanto para el sector privado (en términos de margen bruto) como público (en términos de recaudación). Así pues, como se aprecia en Figura 2.2d, tarifas elevadas generarían importantes pérdidas de eficiencia económica.

Figura 2.2. *Impacto socioeconómico de la tarifación del agua de riego (método MAUF).*



Para terminar este apartado, conviene indicar finalmente que, si bien todas las explotaciones tipo analizadas reproducen el patrón de comportamiento arriba señalado, existen ciertas diferencias de detalle entre las mismas. En este sentido cabe señalar a la explotación tipo 1 (“*Grandes regantes profesionales*”) como aquella que genera un mayor nivel de empleo y una mayor contribución a la recaudación por unidad de superficie, puesto que se trata de los regantes con cultivos más intensivos y de mayor valor añadido. Sin embargo, cuando la tarifa llega a niveles muy elevados (mayores de 0,5 €/m³), haciendo inviables la mayoría de cultivos hortícolas, son igualmente los que provocan una mayor pérdida de eficiencia económica (disminución conjunta del margen bruto y la recaudación). Por su parte, la explotación tipo 3 (“*Agricultores conservadores extensivos*”), a partir de 0,3 €/m^3 deja de regar de manera abrupta (cuando los cultivos de la remolacha, el algodón y el maíz en regadío resultan inviables). No obstante, al tratarse de las explotaciones orientadas hacia las producciones de menor valor añadido, son las que evidencian una menor pérdida de eficiencia económica motivada por tarifas elevadas del agua de riego.

2.4. Discusión

En el presente trabajo se han aplicado tres métodos de PM diferentes para la simulación de impactos de la tarifación del agua de riego. Los resultados obtenidos han sido en buena medida coincidentes, mostrando el mismo patrón de comportamiento para cada explotación tipo analizada y para el conjunto de la zona regable. No obstante, como señalan Howitt *et al*. (1980), para la realización de este tipo de trabajos de simulación resultan más adecuados los métodos de PM no lineales, tal y como la PMP o la MAUF empleados en este trabajo, ya que tales modelos presentan una aproximación más realista al comportamiento de los agricultores (las soluciones óptimas modifican los parámetros definitorios del escenario actual) que la maximización del beneficio mediante programación lineal.

Asimismo, debe señalarse que la similitud de resultados obtenidos entre la PMP y la MAUF está motivada por el perfil comercial de todos los agricultores de la zona analizada (preponderancia del atributo del MBT respecto al resto de atributos en la metodología MAUF en todas las explotaciones tipo), por lo que esta evidencia no puede generalizarse para cualquier otra zona regable. Efectivamente, cuando el comportamiento de los agricultores objeto de estudio pueda considerarse próximo a la maximización del beneficio, PMP y MAUF serán enfoques de PM igualmente válidos para la simulación de los efectos de potenciales escenarios de política agraria, ya que las funciones objetivo a optimizar en ambos casos conducirán a resultados similares. No obstante, como señalan Gómez-Limón y Berbel (2000), Gómez-Limón y Riesgo (2004) o Gutiérrez-Martín y Gómez-Gómez (2011), cuando éste no sea el caso, y existan grupos de agricultores con un comportamiento productivo significativamente diferente de la maximización del beneficio (p. ej., agricultores con alto grado de aversión al riesgo o la complejidad gerencial), parece que el enfoque MAUF sería el más adecuado ya que, a diferencia de la PMP, este enfoque de PM permite considerar el conjunto de criterios de gestión que tales productores tienen en mente a la hora de su toma de decisiones.

El análisis del efecto de la tarifación sobre la demanda de agua y otros indicadores económicos (rentas agrarias) y sociales (generación de empleo) han sido ampliamente estudiados. A continuación se comparan de manera crítica las evidencias reportadas en la literatura y los resultados obtenidos en este trabajo.

En un primer lugar, debe señalarse que la existencia de dos tramos diferenciados en las curvas de demanda de agua de riego, uno primero de carácter inelástico para tarifas bajas y otro elástico para precios más elevados, ya ha sido evidenciada en muchos trabajos realizados anteriormente (Varela-Ortega *et al*., 1998; Gómez-Limón y Berbel, 2000; Gómez-Limón *et al*., 2002; Molle y Berkoff, 2007; Wheeler *et al*., 2008). Así, se encuentra un tramo inicial inelástico, en el que el agricultor conserva su plan de cultivos sin disminuir el uso de agua y un posterior tramo elástico donde se produce una sustitución de los cultivos más intensivos en el uso del agua por otros con menores requerimientos hídricos, incluso sustituyendo cultivos de regadío por otros de secano.

El análisis de la literatura existente y los resultados del presente estudio, evidencian cómo el mayor o menor intervalo de tarifas correspondiente al tramo inelástico depende de la productividad del agua (mayor productividad, mayor tramo inelástico), que en última instancia depende de las ventajas competitivas proporcionadas por las condiciones edafoclimática y el nivel tecnológico presente en la zona regable analizada. Así puede observarse cómo, en sistemas muy extensivos y poco modernizados (riego por superficie), como los sistemas cerealistas del Duero analizados en Gómez-Limón y Berbel (2000), el primer tramo inelástico termina en una tarifa del agua muy baja, inferior a 0,05 €/m^3. Por el contrario, en sistemas más intensivos y tecnificados (riesgo por goteo y aspersión), que incluyen cultivos hortícolas y frutales, como en este trabajo, el tramo inelástico se conserva hasta tarifas mucho mayores, superiores a los 0,25 €/m^3.

En cualquier caso, debe señalarse que la relativa amplitud del tramo inelástico de la curva de demanda obtenido en nuestro caso de estudio puede estar condicionado igualmente por el enfoque cortoplacista seguido en la modelización, ya que éste limita la capacidad de respuesta de los productores frente la introducción de tarifas volumétricas (p. e.j, al no posibilitar cambios en las técnicas de riego). Así, es probable que en un enfoque a más largo plazo, en el que se permitieran cambios en las técnicas de riego, el tramo inelástico se reduciría significativamente (Iglesias y Blanco, 2008).

En segundo lugar, este trabajo evidencia igualmente el impacto diferencial que tiene la tarifación sobre las explotaciones agrarias de una misma zona regable, en función de su orientación productiva y la rentabilidad de sus planes de cultivos. En este sentido, los resultados de este trabajo son coincidentes con las evidencias aportadas por Gómez-Limón y Riesgo (2004). Efectivamente, se puede apreciar cómo dentro de una misma zona regable, incluso en aquellas que emplea las tecnologías de riego más eficientes (Mejías *et al*., 2004), coexisten explotaciones agrícolas con productividades del agua muy diferentes. Por ese motivo, la respuesta de estas frente a la tarifación de estas explotaciones es igualmente dispar.

Así, mientras que las explotaciones con producciones más extensivas (explotación tipo 3 en este trabajo) se muestran más reactivas a los cambios en el precio del agua de riego, modificando rápidamente su plan de cultivos en función de la tarifa aplicable, las explotaciones con planes de cultivos más intensivos (explotación tipo 1 en este trabajo, donde predominan los hortícolas) se muestran menos sensibles a la tarifación, y mantienen su plan de cultivos hasta elevadas tarifas, para las cuales ni siquiera los cultivos de mayor valor añadido son viables económicamente. Esta disparidad en el impacto de la tarifación deja entrever el gran potencial que tendrían los mercados de agua locales (entre regantes de una misma comunidad de regantes), donde los agricultores más conservadores (cultivos de bajo valor añadido) podrían ceder sus derechos en contextos de escasez a los más profesionales (cultivos de mayor valor añadido), tal y como señalan Rey *et al*. (2014) y Calatrava *et al*. (2015).

En tercer lugar, debe señalarse que el análisis de impactos socioeconómicos de la tarifación del agua de riego, en términos de pérdida de renta agraria, empleo y eficiencia económica, también ha sido reportado anteriormente por Varela-Ortega *et al*. (1998), Berbel y Gutiérrez (2006), Riesgo y Gómez-Limón (2006), Iglesias y Blanco (2008), y más recientemente por Kahil *et al*. (2016) y Pérez-Blanco *et al*. (2016). Tal y como se ha comentado en este trabajo, los trabajos antes citados evidencian que el incremento del precio del agua de riego produce cambios en los planes de cultivos que implican siempre pérdidas de rentas agrarias, así como una reducción del empleo directo generado por el sector de regadío, como consecuencia tanto del incremento de costes variables por el pago de la tarifa como por la sustitución de cultivos de elevado valor añadido y consumo de agua por otros menos rentables y con menores necesidades hídricas.

En cuanto a la recaudación pública proveniente de la tarifación, los trabajos anteriores señalan igualmente que, tras el tramo de demanda de agua inelástico, se produce una rápida disminución de la recaudación pública motivada por los drásticos cambios en los planes de cultivos. Efectivamente, cuando las tarifas superan un determinado límite, la demanda entra en un tramo elástico donde la reducción porcentual del consumo de agua es muy superior al incremento porcentual del precio del agua, lo que conduce a una disminución de la recaudación pública.

El comportamiento de la demanda de agua de riego evidenciado en este trabajo y corroborado por buena parte de la literatura existente, sugiere que la tarifación no resulta ser un instrumento tan efectivo como podría pensarse para la gestión tanto en aguas superficiales como subterráneas (Calatrava *et al*., 2011; Kahil *et al*., 2016). Efectivamente, para que la tarifación tenga un efecto disuasorio sobre el consumo, los precios a aplicar deben ser lo suficientemente elevados para alcanzar el tramo elástico. Sin embargo, una vez alcanzado este tramo que permite reducir el consumo, aparecen pérdidas de eficiencia económica, en la medida que las pérdidas de rentas agrarias y de empleo superan con creces la recaudación pública. Así, a pesar de lo que podría asumirse inicialmente en trabajos como el de Tsur *et al*. (2004), que señalan que con la tarifación se podría obtener una distribución más eficiente del agua para el regadío, los resultados obtenidos en este trabajo confirman que la tarifación no puede considerarse como el único instrumento económico dentro de la política de demanda orientada a fomentar un uso del agua en agricultura más racional y sostenible. Efectivamente, la tarifación es considerada menos efectiva que los mercados de agua (Cornish *et al*., 2004) en este sentido, e incluso es considerada como el peor de los instrumentos económicos posibles (Kahil *et al*., 2016), no sólo en términos de beneficios ambientales y privados, sino también en términos de equidad y en cuanto al uso del agua. Por tanto, la tarifación debería considerarse en cualquier caso dentro de un paquete o "mix" de instrumentos económicos, que integre igualmente seguros agrarios (Massarutto, 2015) y mercados de agua. De manera semejante, la desalación también es considerada por otros autores como complemento a la tarifación de agua. En este sentido, Calatrava *et al*. (2011) proponen la combinación de una tasa sobre las extracciones de aguas subterráneas y la sustitución de éstas por agua desalada subvencionada que permita eliminar la sobreexplotación de los acuíferos compatibilizando la contención del coste presupuestario con la minimización del impacto sobre el sector agrario.

Finalmente, cabe comentar que el trabajo realizado debe complementarse en un futuro con nuevas investigaciones encaminadas al análisis del impacto de la tarifación en un contexto temporal más largo, que contemple la posibilidad de cambios estructurales en las explotaciones de regadío (sistemas de riego, tamaño). Sólo de esta manera se podrían cuantificar los impactos económicos, sociales y ambientales de la tarifación en el largo plazo, horizonte en el que se enmarcan los objetivos de DMA, encaminados a la preservación ambiental de las masas de agua y la compatibilidad de sus usos económicos.

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Capítulo 3
Simulating farmers' decision-making with a Cobb-Douglass MAUF. An application for an ex-ante policy analysis of water pricing

Simulating farmers' decision-making with a Cobb-Douglass MAUF. An application for an ex-ante policy analysis of water pricing[2]

Abstract

Classical economic theory relies on the assumption that farmers' behavior can be modeled by maximizing profits or any utility function with profits as a single attribute. However, farmers' decision-making processes are actually driven by various typically conflicting criteria, in addition to the expected profit. Therefore, it must be assumed that producers' behavior is guided by the maximization of a multi-attribute utility function (MAUF) in which all relevant attributes considered for decision-making are condensed. The objective of this paper is to provide more in-depth knowledge about simulating farmers' behavior by using non-linear MAUFs, developing a new non-interactive method to elicit Cobb-Douglas MAUFs based on farmers' actual behavior that overcomes some shortcomings of traditional additive MAUFs. Moreover, this approach is compared with two others that are widely used: the profit maximization and additive MAUF approaches. This procedure is implemented for illustrative purposes to analyze the feasible impacts of water pricing in an irrigated district in southern Spain. The results obtained show that simulations using the Cobb-Douglas utility function are more reliable than the alternatives already used in the literature. In this regard, two pieces of evidence justify this assessment: the calibration is more precise, and the resulting water-demand curve is smoother than in the other two alternative simulation approaches considered.

Keywords: Farmers' behavior, Mathematical programming, Multi-attribute utility theory, Non-linear MAUF, Simulation analysis, Water policy.

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3.1. Introduction

Classical economic theory relies on the assumption that farmers' behavior can be modeled by maximizing profit or any utility function with profit as a single attribute, as assumed by expected utility theory (EUT). In fact, EUT has become one of the most popular approaches to simulating farmers' decision-making, being implemented using several mathematical programming tools (Chavas *et al*., 2010). However, there is large amount of evidence not only supporting the consideration of expected profit for farmers' decision-making processes but also agreeing that these processes are driven by various – typically conflicting – criteria related to their economic, social, cultural, and natural environmental criteria (for recent empirical studies confirming this idea see Berkhout *et al*., 2011; Mandryk *et al*., 2014). Hence, it can be assumed that producers' decision-making is guided by the maximization of a multi-attribute utility function (MAUF), in which all of the relevant attributes considered are condensed. This is the main idea underlying *multi-attribute utility theory* (MAUT), an approach largely developed after the publication of the seminal study by Keeney and Raiffa (1976) to overcome the limitations of single-attribute (profit-related) utility functions. This alternative approach has also been widely implemented in simulating farmers' behavior, as shown by Sumpsi *et al*. (1997), Amador *et al*. (1998), Gómez-Limón and Berbel (2000) and Gómez-Limón *et al*. (2004), among others.

Most empirical approaches to implementing MAUT to simulate farmers' decision-making have relied on additive MAUFs ($U = w_1 \cdot f_1 + w_2 \cdot f_2 + \cdots + w_m \cdot f_m$, where f_1, f_2,… f_m are the different attributes considered and w_1, w_2,… w_m are the weights given by farmers to each attribute) since these linear specifications of the utility function are easier to elicit and to interpret. These MAUFs have typically been estimated using a non-interactive procedure based on weighted goal programming (WGP), as shown by Sumpsi *et al*. (1997), Amador *et al*. (1998) or Gómez-Limón *et al*. (2004). This approach makes it possible to obtain the weights (w_a) of every single attribute with respect the total utility by solving an equation system in which a linear combination of the optimum of each attribute equals the observed attribute levels (for further details, see Section 2.2). However, this additive specification has some shortcomings from an economic perspective, with the most relevant being the assumptions regarding the constant marginal rate of substitution among attributes due to the consideration of linear indifference curves and the total compensation among attributes. This makes additive MAUFs inaccurate when simulating actual decision-making (Hardaker *et al*., 2007). For this reason, in this paper, we propose the elicitation of a Cobb-Douglas utility function ($U = f_1^{\alpha_1} \cdot f_2^{\alpha_2} \cdot \ldots \cdot f_m^{\alpha_m}$), where α_1, α_2,… α_m are parameters related with the relevance given by farmers to each attribute) as a sounder approach, as already suggested by Gutiérrez-Martín and Gómez-Gómez (2011) and Gómez-Limón *et al*. (2016). This choice is justified because this function shape is more coherent with economic theory since it meets conditions of Inada (1963) that guarantee that there is a global optimum when the efficient frontier is convex, and it is consistent with the postulate of decreasing marginal utility for every attribute.

Gutiérrez-Martín and Gómez-Gómez (2011) and Gómez-Limón *et al*. (2016) have recently provided two different non-interactive approaches to eliciting the alpha coefficients (α_a) for Cobb-Douglas MAUFs. In general, the methods developed in both papers are based on the elicitation of the alpha coefficients by equating the marginal rate of transformation between attributes on the efficient frontier and the marginal rate of substitution between attributes on the indifference curves. However, both approaches involve a relatively complex operational burden that makes their implementation in a real-world setting difficult.

The general objective of this paper is to provide more in-depth knowledge about simulating farmers' behavior by using non-linear MAUFs. To that end, this objective is two-fold. First, a new and simpler method is developed to elicit Cobb-Douglas MAUFs based on farmers' actual behavior. Second, this method is implemented for illustrative purposes to simulate farmers' behavior in case a water pricing policy were in place to show that this method is easier to implement than actual simulation exercises. In addition, the results obtained in this manner are compared with those resulting from simulation models that use profit and additive MAUF maximization, confirming the advantages of the Cobb-Douglas MAUF approach for simulation purposes.

To reach the objectives described above, this paper is organized as follows. After this introductory section, the following section introduces the new method developed to elicit Cobb-Douglas MAUFs as a sounder approach compared to existing alternatives to simulating farmers' behavior such as profit or additive MAUF maximization. Section 3 is focused on a real case study considered for the methodological implementation. First, the rationale of irrigation water pricing is explained; second, the irrigation farm type considered for modeling purposes is described. The following section describes the model construction, showing the decision variables, the attributes to be included within the MAUFs, the model constraints and the water price scenarios considered. Section 5 presents the results, both those from the MAUF calibration procedures and those from the simulations of water prices implemented. Finally, in view of the results obtained, Section 6 concludes, reviewing the advantages of the Cobb-Douglas simulation approach and the procedure developed to elicit these MAUFs.

3.2. Simulating farmers' decision-making: alternative approaches

Farmers' decision-making aims to choose productive alternatives (i.e., crop and agricultural technique mixes) that maximize the farmers' utility. This utility can come from a single attribute (profit, according to classic economic theory) or from various attributes (constituted by more complex utility functions, as assumed by MAUT). Regardless of the assumption considered regarding farmers' behavior (mono- or multi-attribute-guided), farmers' utility functions are considered to be a structural feature of these producers; that is, these utility functions do not change when circumstances do. For this reason, these functions can be used to simulate future scenarios by maximizing the corresponding utility function while changing the scenario specific parameters.

Moreover, to simulate farmers' behavior using mathematical programming (MP), the entire set of decision variables (i.e., all feasible crop and agricultural technique mixes) and all physical, technical, market or legal constraints that narrow the set of feasible solutions must be taken into account. In this manner, MP models for simulating farmers' decision-making are built considering an objective function (a mono- or multi-attributed utility function that, in turn, depends on the decision variables) to be maximized and the set of constraints limiting farmers' choices. In this section, we introduce three different approaches to modeling famers' decision-making in this manner, with the last approach being a new contribution to simulating farmers' behavior.

The methodological approaches used in this paper are explained following an evolutionary rationale. First, an MP model relying on a classic mono-attribute objective function with profit as the only relevant attribute for decision-making is introduced.

Considering the multi-criterial nature of farmers' decision-making, the second approach presented is an MP model that maximizes an additive MAUF, elicited following the WGP method proposed by Sumpsi *et al*. (1997). This approach implies linear specifications for MAUFs that involves some shortcomings that warrant discussion (Hardaker *et al*., 2007). Considering additive MAUFs implies linear indifference curves (also called iso-utility curves or iso-preference curves), a condition involving a constant marginal rate of substitution among attributes that leads to oversimplified simulations of farmers' behavior. Furthermore, additive MAUFs allow total compensability among attributes; this is, lower values of every particular attribute can be compensated for by higher values of any other attribute, even if the former reach unacceptably low levels for farmers. This implication also makes additive MAUFs inaccurate when simulating farmers' actual decision-making.

Both limitations of additive MAUFs can be overcame with other utility specifications. In this sense, André and Riesgo (2007) have shown how the application of multiplicative utility functions can be more successful than additive utility functions in reproducing farmers' behavior. For this reason, Gutiérrez-Martín and Gómez-Gómez (2011) Gómez-Limón *et al*. (2016) and have proposed the use of Cobb-Douglas utility functions as a general and flexible multiplicative form for MAUFs that allow more real indifference curves and partial compensation between attributes. Moreover, this function is coherent with neoclassic economic theory since it guarantees that there is a global optimum when the efficient frontier is convex, and this formulation is consistent with the postulate of decreasing marginal utility for every attribute. Thus, the third approach proposed for simulating famers' behavior is an MP model that maximizes a Cobb-Douglas MAUF. As noted above, to that end, a new and simpler procedure for eliciting Cobb-Douglas MAUFs is explained.

3.2.1. Profit maximization

The more classical modeling approach to simulating farmers' behavior is to construct an MP model that considers profit (π) as the unique attribute to be maximized under a set of constraints. Thus, in this case, the model is as follows:

$$Max \;\; U(\boldsymbol{X}) = \pi(\boldsymbol{X}) \tag{1.1}$$

$$\text{s.t.} \qquad \boldsymbol{AX} \leq \boldsymbol{B} \tag{1.2}$$

$$\boldsymbol{X} \geq 0 \tag{1.3}$$

where $\boldsymbol{X}$ ($n \times 1$) is the vector of decision variables (the area devoted to each crop-technique mix), $U(\boldsymbol{X})$ is the objective function (total utility) to be maximized and $\pi(\boldsymbol{X})$ is the profit function. The model constraints are built based on matrix $\boldsymbol{A}$ ($p \times n$) of technical coefficients of the allocable resource constraints and vector $\boldsymbol{B}$ ($p \times 1$) of the available resource levels.

3.2.2. Additive MAUF maximization: WGP approach

The empirical implementation of MAUT for modeling purposes make it necessary to consider some assumptions. The most relevant assumption is that all of the attributes (f_a) contained within the MAUF (U) must be utility-independent[3]. This allows the entire utility $U = U(f_1, \ldots f_a, \ldots f_m)$ to became a separable function: $U = g[u_1(f_1), u_2(f_2), \ldots u_m(f_m)]$. Moreover, if both total and partial utility functions take values in the range of zero to one, then the MAUF takes either the additive form ($U = \sum w_a \cdot u_a(f_a)$) or the multiplicative form ($U = [\prod(K \cdot w_a \cdot u_a(f_a) + 1) - 1]/K$), where $0 \leq w_a \leq 1$ and $K = f(w_a)$. If attributes are mutually utility-independent and $\sum w_a = 1$, then $K = 0$, and the utility function is additive. By contrast, if $\sum w_a \neq 1$, then $K \neq 0$, and the mathematical form is multiplicative (Fishburn, 1982).

[3] According to Keeney and Raiffa (1976), attribute i is defined as the utility independent of attribute j when the conditional preferences for lotteries on attribute i given the attribute j do not depend on the particular level of attribute j.

Considering that farmers' decision-making is guided by m attributes f_a and the abovementioned requirements for an additive MAUF are fulfilled, the simulation MP model can be specified as follows:

$$Max\ \ U(\boldsymbol{X}) = \sum_{a=1}^{m} w_a \cdot u_a\big(f_a(\boldsymbol{X})\big) = \sum_{a=1}^{m} w_a \cdot nf_a(\boldsymbol{X}) \tag{2.1}$$

$$\text{s.t.} \qquad \sum_{a=1}^{m} w_a = 1 \tag{2.2}$$

$$\boldsymbol{AX} \leq \boldsymbol{B} \tag{2.3}$$

$$\boldsymbol{X} \geq 0 \tag{2.4}$$

This objective function depends on a set of m single or partial utility functions $(u_a\big(f_a(\boldsymbol{X})\big))$ that consider all relevant attributes for producers' decision-making, and wa denotes the weight of each attribute, expressing its relative importance.

For operational and comparability purposes, we consider that all relevant attributes are related to objectives to be maximized (i.e., more-is-better attributes). This assumption does not imply any loss of generality. A less-is-better attribute (objective to be minimized) can be transformed into a more-is-better attribute simply by multiplying it by -1. If the attribute is to precisely reach a certain target (goal), then it can be written as an objective minimizing the distance (or maximizing the opposite of the distance) from the attained value to the target value, so that it can be formulated as a less-is-better (or more-is-better) objective. Therefore, the formulation proposed, which considers all attributes as objectives to be maximized, allows us to address any problem involving any of the relevant types of attributes (objectives or goal types) considered in the farmer's MAUF.

Moreover, it is assumed that each single-attribute or partial utility function $(u_a\big(f_a(\boldsymbol{X})\big))$ is equal to the corresponding attribute $f_a(\boldsymbol{X})$. This assumption implies linear utility-indifferent curves (constant partial marginal utility), a somewhat strong assumption that can be regarded as a close enough approximation if the attributes vary within a constrained range (Hardaker *et al*., 2007). Huirne and Hardaker (1998) show how the slope of the single-attribute utility function has little impact on the ranking of alternatives. Similarly, Amador *et al*. (1998) analyze how linear and quasiconcave functions yield almost the same results. Consequently, we assume this simplification in the elicitation of the MAUFs. Finally, for operational purposes, the attribute functions are properly normalized to be bounded between 0 and 1 ($nf_a(\boldsymbol{X})$).

Sumpsi *et al*. (1997) describe a widely used non-interactive process for eliciting the values of the calibrating parameters wa in additive MAUFs. Following these authors, the crop mix selection ($\boldsymbol{X}$) can be viewed as a multi-objective programming (MOP) decision-making problem. Because the preferences of decision-makers should belong to the efficient frontier, a first approximation can be assessed through the pay-off matrix, which is obtained by maximizing each of the objectives (in our case, partial utility functions, $nf_a(\boldsymbol{X})$) separately, subject to the constraints set (expressions 2.2 to 2.4). To obtain the relative weight of each attribute (w_a), a system of equations to make the sum of the weighted elements of the pay-off matrix for each attribute be equal to their observed value (nf_a^{obs}) is built as follows:

$$\begin{bmatrix} nf_{11} = nf_1^* & nf_{12} & \cdots & nf_{1m} \\ nf_{21} & nf_{22} = nf_2^* & \cdots & nf_{2m} \\ \vdots & \vdots & \ddots & \vdots \\ nf_{m1} & nf_{m2} & \cdots & nf_{mm} = nf_m^* \end{bmatrix} \cdot \begin{bmatrix} w_1 \\ w_2 \\ \vdots \\ w_m \end{bmatrix} = \begin{bmatrix} nf_1^{obs} \\ nf_2^{obs} \\ \vdots \\ nf_m^{obs} \end{bmatrix} \tag{3.1}$$

$$\sum_{a=1}^{m} w_a = 1 \tag{3.2}$$

where nf_a^* is the normalized ideal value for attribute a, $nf_{aa'}$ are the normalized values of the elements in the pay-off matrix of attribute a when attribute a' is optimized, and nf_a^{obs} are the normalized observed values of each attribute.

Typically, however, there is not an exact solution to the above system; thus, it is necessary to solve the problem by minimizing the sum of the deviational variables that determine the set of w_a that provided the closest solution:

$$Min \sum (n_a + p_a) \tag{4.1}$$

s.t. $$w_1 \cdot nf_{11} + w_2 \cdot nf_{12} + \cdots + w_m \cdot nf_{1m} + n_1 - p_1 = nf_1^{obs} \tag{4.2}$$

$$w_1 \cdot nf_{21} + w_2 \cdot nf_{22} + \cdots + w_m \cdot nf_{2m} + n_2 - p_2 = nf_2^{obs} \tag{4.3}$$

$$\cdots \qquad \cdots$$

$$w_1 \cdot nf_{m1} + w_2 \cdot nf_{m2} + \cdots + w_m \cdot nf_{mm} + n_m - p_m = nf_m^{obs} \quad (4.\text{m}+1)$$

$$w_1 + w_2 + \cdots + w_m = 1 \quad (4.\text{m}+2)$$

where n_a and p_a are the negative and positive deviations from the observed values for each attribute, respectively.

3.2.3. Cobb-Douglas MAUF maximization: WGP approach

Considering a Cobb-Douglas MAUF, the MP model proposed for simulating farmers' behavior takes the following form:

$$Max\ \ U(\boldsymbol{X}) = \prod_{a=1}^{m}\left[u_a\big(f_a(\boldsymbol{X})\big)^{\alpha_a}\right] = \prod_{a=1}^{m}\left[nf_a(\boldsymbol{X})^{\alpha_a}\right] \quad (5.1)$$

$$\text{s.t.} \qquad \sum_{a=1}^{m} \alpha_a = 1 \quad (5.2)$$

$$\boldsymbol{AX} \leq \boldsymbol{B} \quad (5.3)$$

$$\boldsymbol{X} \geq 0 \quad (5.4)$$

where α_a denotes the calibration coefficients of each attribute, related to their relative importance.

As established above for the additive MAUF, in the case of the Cobb-Douglas specification, it is also assumed that all attributes are related to objectives to be maximized (i.e., more-is-better), and each single-attribute or partial utility function is equal to the corresponding attribute properly normalized to be bounded between 0 and 1. For purposes of comparability, the same normalization has been performed in all approaches, including profit maximization.

The new methodological approach proposed for eliciting the α_a coefficients in Cobb-Douglas MAUFs is also based on the WGP approach explained above. To that end, first, the Cobb-Douglas function is transformed into an additive expression. In mathematical terms, one of the advantages of the Cobb-Douglas function used as an objective function (expression 5.1) is the possibility of being transformed into an additive function using logarithms without losing any of its features:

$$log[U(\boldsymbol{X})] = V(\boldsymbol{X}) = \sum_{a=1}^{m} \alpha_a \cdot log[nf_a(\boldsymbol{X})] \tag{6}$$

Following a procedure similar to that developed by Sumpsi *et al*. (1997), this transformation makes it possible to estimate the most appropriate alpha parameters by solving the following m+1 system of equations, in which the weighted sum of the elements of the pay-off matrix are equal to the observed values of the attributes, all of them properly normalized and transformed by the natural logarithms:

$$\begin{bmatrix} log(nf_{11}) = \log(nf_1^*) & log(nf_{12}) & \cdots & log(nf_{1m}) \\ log(nf_{21}) & log(nf_{22}) = log(nf_2^*) & \cdots & log(nf_{2m}) \\ \vdots & \vdots & \ddots & \vdots \\ log(nf_{m1}) & log(nf_{m2}) & \cdots & log(nf_{mm}) = log(nf_m^*) \end{bmatrix} \cdot \begin{bmatrix} \alpha_1 \\ \alpha_2 \\ \vdots \\ \alpha_m \end{bmatrix} = \begin{bmatrix} log(nf_1^{obs}) \\ log(nf_2^{obs}) \\ \vdots \\ log(nf_m^{obs}) \end{bmatrix} \tag{7.1}$$

$$\sum_{a=1}^{m} \alpha_a = 1 \tag{7.2}$$

As in the additive WGP approach, the previous system may not have an exact solution (this is typically case). Therefore, it is necessary to solve the problem by minimizing the sum of deviational variables that determine the set of alpha parameters that lead to the closest solution:

$$Min \sum_{a=1}^{m} (n_a + p_a) \tag{8.1}$$

s.t. $$\alpha_1 \cdot log(nf_{11}) + \alpha_2 \cdot log(nf_{12}) + \cdots + \alpha_m \cdot log(nf_{1m}) + n_1 - p_1 = log(nf_1^{obs}) \tag{8.2}$$

$$\alpha_1 \cdot log(nf_{21}) + \alpha_2 \cdot log(nf_{22}) + \cdots + \alpha_m \cdot log(nf_{2m}) + n_2 - p_2 = log(nf_2^{obs}) \tag{8.3}$$

$$\cdots \qquad \cdots$$

$$\alpha_1 \cdot log(nf_{m1}) + \alpha_2 \cdot log(nf_{m2}) + \cdots + \alpha_m \cdot log(nf_{mm}) + n_m - p_m = log(nf_m^{obs}) \tag{8.m+1}$$

$$\sum_{a=1}^{m} \alpha_a = 1 \qquad (8.m+2)$$

where n_a and p_a are the absolute negative and positive deviations, respectively.

Once the alpha parameters are estimated by running model (8), the shape of the Cobb-Douglas MAUF to be used for modeling purposes is as follows:

$$U(\boldsymbol{X}) = nf_1(\boldsymbol{X})^{\alpha_1} \cdot nf_2(\boldsymbol{X})^{\alpha_2} \cdot \ldots \cdot nf_m(\boldsymbol{X})^{\alpha_m} \qquad (9)$$

3.3. Case study

3.3.1. Irrigation water pricing

At present, water resources are increasingly scarcer in Spain because of rising demand and declining availability due to climate change. Moreover, traditional supply-side water policy instruments, such as the construction of dams and other water infrastructure to increase water supply, cannot be further developed since new increases in the water supply are technically infeasible or economically unaffordable, a situation known as 'basin closure' (Molle *et al*., 2010). When basin development reaches the closure stage, any new water demand must be satisfied by reducing other existing water uses. Under these circumstances, demand-side water policy instruments such as water pricing or water markets are considered the most suitable solutions for allowing a more efficient reallocation of water resources (Lago *et al*., 2015).

Closed basins are found not only in Spain but also in other member states of the European Union (EU) and other countries worldwide. This situation has caused EU institutions to decide to develop a common policy for water management. The approval of the Water Framework Directive (WFD; Directive 2000/60/CE of the European Parliament and of the Council) it is considered the main achievement in this field (Kallis and Butler, 2001). The WFD (article 9) proposes water pricing as the main policy instrument for addressing the demand for water within the EU (European Commission, 2001).

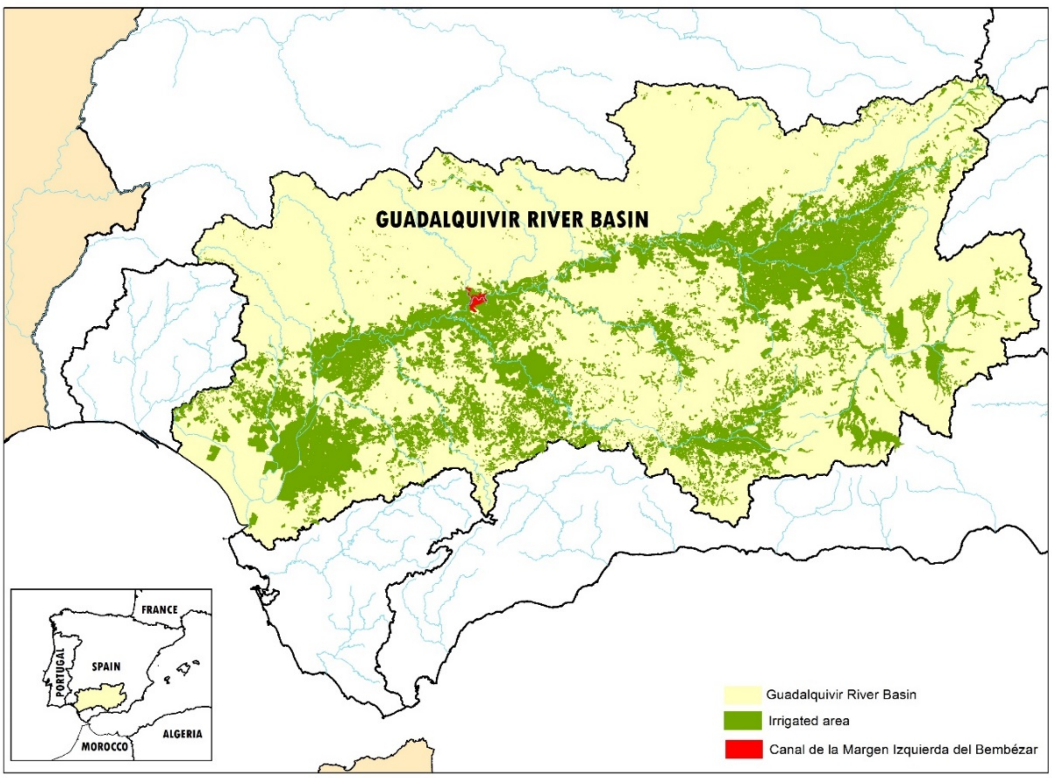
The rationale on which water pricing is based is purely economic. In this sense, farmers in irrigated areas, according to economic theory, will respond to the introduction of (or an increase in) water prices by reducing their consumption, in accordance with a negatively sloped demand curve. In this manner, the water savings obtained would be re-distributed among other uses such as productive or environmental purposes (ecological flows in rivers, etc.), according to societal preferences. Such a reallocation of water resources would improve the efficiency of their use (Johansson *et al*., 2002; Tsur *et al*., 2004). The impact of water pricing on farmers' behavior has been widely studied in the literature. In this sense, it is worth noting the book edited by Dinar *et al*. (2015), who show the experiences of water pricing in several countries around the world (Australia, Brazil, Canada, Chile, China, Colombia, France, India, Israel, Mexico, The Netherlands, New Zealand and South Africa). Also relevant is the book edited by in Berbel and Gutiérrez-Martín (2004), where interested readers can find a detailed analysis of the impacts of water pricing on irritated agriculture in the European Union, considering case studies in Spain, Italy, Greece and Portugal. Other interesting works addressing water pricing in European countries are the ones developed by Bontemps and Couture (2002) in France, focused on the estimation of water demand when water pricing is implemented; Manos *et al*. (2006) in Greece and Fragoso and Marques (2015) in Portugal, where the impacts of water pricing under alternative scenarios of European agricultural policy are analyzed; and Bartolini *et al*. (2007), Galioto *et al*. (2013) and Pérez-Blanco *et al*. (2016) in Italy, regarding the design of tariff strategies aiming at cost recovery. Likewise, recent empirical studies by Pérez-Blanco *et al*. (2015), Kahil *et al*. (2016) and Montilla-López *et al*. (2017) about irrigation water pricing in Spain are also worth to be cited.

Most empirical studies that analyze how farmers would react in case water prices were introduced (or increased) have relied on MP models to simulate the feasible behavior of these producers when the parameter in the model representing the volumetric water price is increased. In this paper, a similar ex-ante policy analysis of water pricing is implemented for a real case study. However, as noted above, this analysis is performed by using three different modeling approaches to illustrate the outperformance of the new method proposed based on a Cobb-Douglas MAUF.

3.3.2. Modeling scope

The empirical application proposed as the case study is developed in the *Canal de la Margen Izquierda del Bembézar* irrigation district, located in the Guadalquivir River Basin in southern Spain (see Figure 3.1). This irrigated area covers a total of 4,009 hectares divided into 163 farms with an average farm size of 66.2 ha.

Figure 3.1. *Location of the selected irrigated area*



Source: own elaboration

Within the same agricultural system (e.g., an irrigation district), it can be easily assumed that all farms fulfill the following features: i) technological homogeneity (the same possibilities of production, the same types of resources, the same technological level and the same management capacity); ii) pecunious proportionality (proportional profit expectations for each activity); and iii) institutional proportionality (the availability of resources to the individual farm proportional to the average availability). Assuming a profit maximizing behavior, if the abovementioned requirements are met, then all farms can be modeled together within the same MP model without aggregation-biased results since all of them are assumed to have a similar productive behavior (Hazell and Norton, 1986). However, real-world observations show that even within the same agricultural system, there exists heterogeneity between farms regarding the crop mixes and agricultural practices (in our case, irrigation techniques), mainly because of the differences in farmers' utility functions since profit maximization is only seldom the unique objective guiding farmers' decision-making (Pennings and Leuthold, 2000; Berkhout *et al*., 2011; Karali *et al*., 2013). In fact, if an MAUT perspective is being considered, an additional homogeneity requirement is needed to avoid aggregation bias, i.e., homogeneity related to the MAUF shape (i.e., the values of the parameters defining additive or Cobb-Douglas MAUFs).

The MAUF shape is primarily based on the psychological characteristics of the decision-makers, which differ significantly from farmer to farmer. According to this perspective, the differences in decision-making (crop mix) among farmers in the same production area must be primarily due to the differences in their utility functions (in which the relative importance given to different criteria are condensed), rather than other differences related to the profits of economic activities or disparities in resource requirements or endowments. Thus, to avoid aggregation bias resulting from lumping together farmers with significantly different MAUFs, a classification of farmers into homogeneous groups with similar decision-making behavior (utility functions) is required.

For this type of classification, the most efficient method is cluster analysis (Berbel and Rodríguez-Ocaña, 1998; Gómez-Limón and Riesgo, 2004; Berkhout *et al*., 2011), taking farmers' real decision-making vectors (the actual crop mix) as the classification criterion. Thus, following this idea, in this paper, we use clustering techniques to assign individual farms to homogenous groups regarding their crop mixes (i.e., the MAUF shape). Within the possibilities that this technique contains, we have selected Ward's procedure as a criterion for aggregation and the Euclidean distance as a measure of the distance between farms of the irrigation district selected as the case study. Following this procedure, a dendrogram has been generated, clearing showing three different clusters; their average profiles (considering both crop mixes as classification variables and other structural variables such as farm size, farmer age, etc.) have been used to define the corresponding 'farm types', as shown in Table 3.1.

Table 3.1. *Farm types in Canal de la Margen Izquierda del Bembézar*

	Label	*Crop mix*	*Farm size (hectares)*	*Agricultural income / Total income (%)*
Cluster 1	Large diversified professional farmers	Corn-Drip (32%), Orange-Drip (25%), Olive-Drip (12%), Sunflower-Sprinkler (8%), Wheat-Sprinkler (6%), Cotton-Sprinkler (6%), Vegetables-Sprinkler (6%), Potato-Sprinkler (3%)	79.7	83.6
Cluster 2	Citrus growers	Orange-Drip (100%)	47.0	65.5
Cluster 3	Small part-time corn growers	Corn-Drip (100%)	13.1	42.9

The homogeneous farms included in each cluster and represented by their own farm type can be properly modeled without aggregation biases. Thus, these farm types are considered decision units to be modeled in the individual MP models. Regardless, considering the illustrative purpose of this paper, from this point on, only the case of Cluster 1 (large diversified professional farmers) is considered for model building.

3.4. Model building

3.4.1. Variables, attributes and objective functions

The decision variables for the farmer are the area devoted to each alternative productive activity ($\boldsymbol{X}$). These activities are denoted as $X_{i,j}$, where i means the crop and j the irrigation technique used. The combination of crops and the irrigation techniques considered as the decision variables for the case study analyzed includes the current irrigated activities shown in Table 3.1 and rain-fed alternatives (wheat, sunflower and olive groves). Thus, the model constructed will be able to simulate the impacts of the various pricing scenarios (the water-demand function) as the result of the famers' short-term production adjustments, simulating both the substitution of water-intensive crops by others and the cessation of irrigation and the introduction of rain-fed crops with no need for water.

For the elicitation of the utility functions, three attributes have been used as the most relevant attributes to model farmers' decision-making, considering the existing evidence (e.g., Gómez-Limón and Riesgo, 2004; Pérez-Blanco and Gutiérrez-Martín, 2017). These attributes are i) the profit in the short run, ii) the risk inherent to this profit, and iii) the managerial complexity associated with the crop mix. Attributes are defined as a mathematical function of the decision variables and become objectives when the direction of improvement of each attribute is set. That is, the objective related to each attribute will be profit maximization, risk minimization and managerial complexity minimization. Profit is defined by the expected total gross margin (the average value of the 2007-2013 times series) ($f_1(\boldsymbol{X}) = GM(\boldsymbol{X})$). Risk is measured as the variance of the gross margin in the same period ($f_2(\boldsymbol{X}) = VAR(\boldsymbol{X})$). Finally, total labor ($f_3(\boldsymbol{X}) = TL(\boldsymbol{X})$) has been selected as a proxy for managerial complexity.

The expected gross margin ($GM(\boldsymbol{X})$) has been calculated as the sum of total income (the average crop price – p_i – multiplied by the average yield – $y_{i,j}$ – plus coupled subsidies – s_i) minus the variable costs ($vc_{i,j}$) and the water cost from the water pricing, which is the product of the water quantity used ($wq_{i,j}$) and the water price (wp):

$$GM(\boldsymbol{X}) = \sum_i \sum_j \{(p_i \cdot y_{i,j} + s_i - vc_{i,j} - wq_{i,j} \cdot wp) \cdot X_{i,j}\} \tag{10}$$

The variance of the gross margin in the time series considered ($VAR(\boldsymbol{X})$) is defined by equation (11), where $\boldsymbol{X}^t$ is the transposed vector $\boldsymbol{X}$ and $[\boldsymbol{cov}]$ is the variance-covariance matrix of the gross margins of productive activities per hectare during the 2007-2013 period.

$$VAR(\boldsymbol{X}) = \boldsymbol{X}^t \cdot [\boldsymbol{cov}] \cdot \boldsymbol{X} \tag{11}$$

Total labor is calculated as shown in equation (12), that is, as the sum of labor requirements per crop and the irrigation technique ($tl_{i,j}$) in the entire farm area.

$$TL(\boldsymbol{X}) = \sum_i \sum_j tl_{i,j} \cdot X_{i,j} \tag{12}$$

As noted above, the objectives related to the several attributes considered (partial utility functions) must be normalized for operational purposes to transform them into more-is-better and dimensionless functions (nf_a), whose values vary within the interval [0,1]. To that end, we propose transforming the original attribute functions into rates of success with respect to the ideal value of each attribute as follows:

$$nf_{GM}(\boldsymbol{X}) = \frac{GM(\boldsymbol{X})}{GM^*}; \quad nf_{VAR}(\boldsymbol{X}) = \frac{VAR^*}{VAR(\boldsymbol{X})}; \quad nf_{TL}(\boldsymbol{X}) = \frac{TL^*}{TL(\boldsymbol{X})} \tag{13}$$

where GM^*, VAR^* and TL^* are the optimal or ideal values for the gross margin, variance and total labor (their maximum and minimum, respectively). Note that whereas for the more-is-better attribute ($GM(\boldsymbol{X})$), the ideal value (the largest possible value) is in the denominator, for the less-is-better attributes ($VAR(\boldsymbol{X})$ and $TL(\boldsymbol{X})$), the ideal values (the smallest possible values) are in the numerator. Thus, it can be checked that operating in this manner, all normalized attributes (nf_a) are related to more-is-better objectives and that their values range between 0 and 1.

The values of the attributes, properly normalized, represent the partial utilities, which are combined in each utility function (objective functions in the MP models) according to each methodological approach described in Section 2. In this regard, equations 14 to 16 represent the objective function in the case of profit maximization (14), the additive MAUF (15) and the Cobb-Douglas MAUF (16):

$$Max\ U(\boldsymbol{X}) = nf_{GM}(\boldsymbol{X}) \tag{14}$$

$$Max\ U(\boldsymbol{X}) = w_{GM} \cdot nf_{GM}(\boldsymbol{X}) + w_{VAR} \cdot nf_{VAR}(\boldsymbol{X}) + w_{TL} \cdot nf_{TL}(\boldsymbol{X}) \tag{15}$$

$$Max\ U(\boldsymbol{X}) = nf_{GM}(\boldsymbol{X})^{\alpha_{GM}} \cdot nf_{VAR}(\boldsymbol{X})^{\alpha_{VAR}} \cdot nf_{TL}(\boldsymbol{X})^{\alpha_{TL}} \tag{16}$$

Note that the normalization of $GM(\boldsymbol{X})$ in (14) is not necessary but has been performed for the sake of homogeneity.

3.4.2. Model constraints

Farmers' decision-making is subject to constraints that limit the feasible set of choices at hand. These constraints respond not only to the fact that resources are limited but also to other restrictions such as crop rotations, agricultural policy quotas, marketing channel limits, etc. The constraints limit the space of the solutions of the model to those that are attainable by the farmers (feasible solution set), explaining a large share of its behavior. Equations 17.1 to 17.4 show all restrictions taken into account by the farmers analyzed:

$$\sum_i \sum_j X_{i,j} \leq fa \tag{17.1}$$

$$\sum_i \sum_j X_{i,j} \cdot wq_{i,j} \leq wa \cdot fa \tag{17.2}$$

$$\boldsymbol{AX} \leq \boldsymbol{B} \tag{17.3}$$

$$X_{i,j} \geq 0; \qquad \forall\, i,j \tag{17.4}$$

Constraints (17.1) and (17.2) are related to land and water availability, respectively. The first limits the total area covered by the different alternatives to the farm size (fa). The water constraint establishes that irrigation water requirements cannot exceed water availability, with the former being the sum of water requirements per alternative and the latter the water allotment provided by the water agency considering farm size (fa) and water rights granted per hectare (wa). Moreover, equation (17.3) denotes the rest of the constraints defining the feasible solution set, which constitute technical (agronomic and irrigation technology), policy (cotton quota) and market requirements:

a) *Agronomic constraints*. These include the rotational and frequency constraints actually followed by farmers as good agricultural practices.

b) *Permanent crops*. In the short run, it is not possible to increase or decrease the area devoted to permanent crops (in our case study, citrus and olive groves) because they are fixed assets that are only changeable in the long run. For this reason, permanent crops are not allowed to change. However, irrigated olive groves are allowed to change into rain-fed groves since this woody crop can be grown with and without irrigation [4].

c) *Irrigation technique*. It is assumed that the specific equipment for each irrigation system (surface, sprinkler and drip) remains the same in the short term (new investments in irrigation technology are not considered). As a result, the maximum area irrigated by each of these systems is fixed. This fact is modeled by preventing the area covered by each irrigation technique from increasing by more than 5% compared to the observed values.

d) *Cotton quota*. The area devoted to cotton is limited to the maximum area observed in the period considered due to an agricultural policy constraint.

e) *Market constraints*. There are crops such as garlic and onions that are subject to limited marketing channels because they cannot be stored for extended periods (perishable products). The implication is that it is unlikely that farmers will significantly increase the area cultivated with such crops due to the inability of the market to absorb great variations in production in the short run. Thus, to model this constraint, an upper limit of the area cultivated with these crops was included on the basis of the maximum historical cultivation during the previous seven years.

Finally, decision variables ($X_{i,j}$) are fixed as non-negative, as denoted by equation (17.4).

3.4.3. *Simulating water pricing*

Simulations can be performed because the utility functions are considered a structural feature of farmers that does not change over the course of any simulation. Thus, the models built as explained in the two previous sections have been used to simulate farmers' responses (in terms of the crop mix and water use) to an increment in the price of water (the parametrization of a volumetric water price affecting farming costs)[5].

[4] This is not possible with orange groves since this permanent crop can be grown only under irrigation.
[5] The current water cost is already included in the variable costs ($vc_{i,j}$)

The parametrization of the water price (wp) allows the water-demand curve to be built; that is, the model results will show the water quantity that farmers are willing to use at every simulated water price. Economic theory assumes that any increment in the water price will lead to a reduction in water use in accordance with a negatively sloped water curve. However, this change in the water quantity used will depend on the elasticity of the demand curve, which is related to its slope.

To perform the methodological comparison proposed by the estimation of the water-demand curves, we have parametrized the water price from €0.00/m^3 to €0.30/m^3 to simulate farmers' decision-making in the short run (only changing the crop mix) using the three approaches described in Section 2.

Finally, it must be noted that simulating the impact of water pricing using MP models will also make it possible to estimate a series of indicators of interest for policy decision-makers, covering economic (e.g., the aggregated gross margin), social (e.g., the aggregated agricultural labor demand) and environmental (e.g., the aggregated agrochemical use) issues (see for example the works by Gómez-Limón and Riesgo, 2004; Gallego-Ayala *et al.*, 2011). In fact, to support policy design and implementation, this type of ex-ante policy evaluation is very useful. However, considering the methodological main purpose of this paper, this policy analysis falls beyond its scope and is thus not reported in this chapter.

3.5. Results and discussion

3.5.1. MAUF calibration and validation of simulation models

Running the calibration procedures as explained in model (4) for the additive MAUF and model (8) for the Cobb-Douglas MAUF, both sets of calibration parameters (weights – w_a – and alphas – α_a –, respectively) are obtained. This allows the elicitation of the objective functions in MP simulation models (2) and (5) built for the farm type considered. Thus, the formulation of expressions (2.1) and (5.1) became as follows:

$$U(\boldsymbol{X}) = 0.89 \cdot nGM(\boldsymbol{X}) + 0.06 \cdot nVAR(\boldsymbol{X}) + 0.04 \cdot nTL(\boldsymbol{X}) \qquad (18)$$

$$U(\boldsymbol{X}) = nGM(\boldsymbol{X})^{0.89} \cdot nVAR(\boldsymbol{X})^{0.11} \tag{19}$$

In the case of the additive MAUF, all proposed attributes are included in the calibrated utility function. However, in the case of the Cobb-Douglas MAUF, only the expected gross margin and variance are taken into account, showing that the contribution of total labor to the total utility is negligible when this utility function is considered. Although weights and alpha parameters are not totally comparable (the alpha parameters are not exactly weights but, rather, a proxy), both approaches show the much greater relevance of the expected gross margin over the rest of the attributes in the decision-making process. As a consequence of this fact, for this case study, the solutions when maximizing both MAUFs will not be too far away from the solution to the first simulation approach (MP model (1)), in which only the expected gross margin is maximized.

To validate the simulation models built for the farm type considered, we proceed to compare the actual situation (observed levels) with the simulated results for the current scenario (Qureshi *et al.*, 1999). These simulations for the current scenario are obtained by maximizing every objective function subject to the constraints considered, as shown in models (1), (2) and (5) for profit maximization, the additive MAUF and the Cobb-Douglas MAUF, respectively. To validate these models, the simulated results obtained in the space of attributes ($GM(\boldsymbol{X})$, $VAR(\boldsymbol{X})$, and $TL(\boldsymbol{X})$) and in the space of decision variables ($\boldsymbol{X}$) are compared.

The results of the comparison between the simulated results for the attributes under the current scenario with those related to the actual crop pattern are shown in Table 3.2. The last row of this table shows the values achieved by the mean squared error (MSE) for each simulation approach. This statistical indicator measures the average of the squares of the errors or deviations between the estimator and what is estimated between the observed and the simulated vectors of the attributes, following the formula below:

$$MSE = \sqrt{\frac{\sum_{a=1}^{m} \left(\frac{f_a^{obs}(\boldsymbol{X}) - f_a(\boldsymbol{X})}{f_a^{obs}(\boldsymbol{X})} \right)^2}{m}} \tag{20}$$

This error aims to clarify which approach best approximates the observed attribute levels. In this regard, the MSE shows that the calibration with the Cobb-Douglas MAUF is the most accurate approach. In other words, the values of the attributes from the simulation using the Cobb-Douglas MAUF approach are closer to the actual values than those from the other approaches.

Table 3.2. *Model validation: Attributes values*

	Profit maximization	*Additive MAUF*	*Cobb-Douglas MAUF*	*Observed*
GM (€/ha)	2,215.85	2,214.69	2,210.53	2,103.19
Risk (€^{2}/ha)	6,598.47	6,510,35	6,316.97	6,079.75
Labor (h/ha)	87.40	87.16	86.32	83.75
Mean Squared Error (MSE)	6.3%	5.6%	4.1%	

Additionally, the validation in the space of the decision variables attempts to analyze the capacity of the model to reproduce the farmers' actual crop mix. To that end, we have calculated two indicators: i) the percentage absolute deviation (PAD) and ii) the Finger-Kreinin similarity index (FK, see Finger and Kreinin, 1979), which are calculated as follows:

$$PAD\ index\ (\%) = \frac{\sum_i \sum_j \left| X_{i,j}^{obs} - X_{i,j} \right|}{fa} \tag{21}$$

$$FK\ similarity\ index = \sum_i \sum_j min\left(\frac{X_{i,j}}{fa}; \frac{X_{i,j}^{obs}}{fa}\right) \tag{22}$$

The PAD compares simulated and observed crop areas by adding all absolute deviations and expressing this summation in perceptual terms. Thus, this index can vary from 0% (perfect calibration fitting) to 200% (the worst possible calibration). Similarly, the FK similarity index compares the simulated and the observed shares of each crop mix, varying between 0% and 100%, with the latter being an exact match between the observed and the simulated crop mixes.

Table 3.3 shows the simulated crop mix for each approach and the observed levels of the different productive alternatives. Additionally, in the last two rows, the two similarity indexes calculated for every approach are presented.

Table 3.3. *Model validation: Decision variables (crop areas in hectares)*

Crop mix	*Profit maximization*	*Additive MAUF*	*Cobb-Douglas MAUF*	*Observed*
Durum wheat-Sprinkler	0.00	0.80	3.65	6.14
Corn-Drip	27.14	27.14	27.14	25.85
Potato-Sprinkler	13.25	12.45	9.60	2.69
Cotton-Sprinkler	4.98	4.98	4.98	4.98
Sunflower-Sprinkler	0.00	0.00	0.00	6.34
Garlic-Sprinkler	1.79	1.79	1.79	1.70
Onion-Sprinkler	3.27	3.27	3.27	2.73
Orange-Drip	19.73	19.73	19.73	19.73
Olive-Drip	9.49	9.49	9.49	9.49
PAD	31.3%	29.3%	22.2%	
FK index	84.3%	85.3%	88.9%	

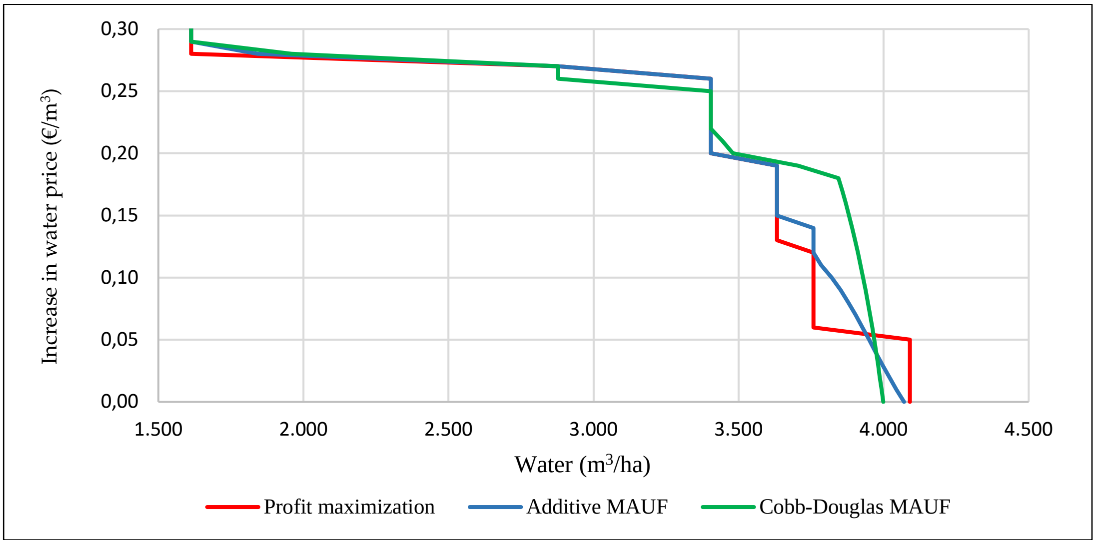
According to these results, it is shown that the most accurate approach is the Cobb-Douglas MAUF since it reaches the lowest PAD (22.2%) and the highest FK index (88.9%). Thus, it is demonstrated that the Cobb-Douglas MAUF approach is once again the best among the approaches considered since for the reference scenario, it reproduces the farmers' behavior better than the other approaches. The additive MAUF approach is ranked second following these two indicators, also outperforming the profit maximization approach, which obtains the worst values in both the PAD and the FK index.

Finally, it is worth noting that since the deviations in the objectives and in the decision variable spaces are sufficiently small in the case of the Cobb-Douglas MAUF approach, it can be affirmed that this modeling approach is a reasonably accurate enough approximation for simulating farmers' actual decision-making.

3.5.2. Demand curves

After the calibrations, the resulting MP simulation models have been run to parametrize the water price from €0.00/m³ to €0.30/m³. From each iteration, the resulting water use has been recovered to construct the different demand curves. These curves show the changing behavior of farmers when an increase in water pricing is implemented, as shown in Figure 3.2.

Figure 3.2. *Water demand considering profit maximization, the additive MAUF and the Cobb-Douglas MAUF approaches*



The first result worth noting is that the shapes of the three curves are somewhat similar, presenting a common inelastic segment (high slope) for low increases in the water price (less than €0.20/m³); this is, relative high increases in the water price lead to relative low decreases in water use. This simulated behavior with a large initial inelastic segment can also be found in many previous empirical studies (e.g., Molle and Berkoff, 2007; Wheeler *et al*., 2008; Montilla-López *et al*., 2017). From €0.20/m³ on, the results obtained for all methods are almost the same.

These similarities in the three demand curves can be explained because of the great relevance of the attribute expected gross margin in this case study, which leads to very similar utility functions in all of the approaches tested. In case the farmers analyzed were more risk or managerial complexity adverse, the results would be greatly different.

Nevertheless, the demand curves show that the differences in the inelastic segments are worth taking into account, with the smoothness of these curves being the most relevant difference. As is widely known, simulations that use the profit maximization approach lead to an inertia in the vicinity of the reference situation and a 'jumpy' behavior that does not make this approach sufficiently reliable (Mérel and Howitt, 2014). MAUF approaches, namely, the new method based on the Cobb-Douglass MAUF, provide much more credible simulation results, avoiding overreactions to exogenous shocks, as the policy change proposed. In fact, the Cobb-Douglass MAUF approach shows the smoothest demand curve, which is known to be a good indicator of realism (Heckelei and Britz, 2005).

3.6. Conclusions

The main contribution of this paper is the development of a new and simpler method to elicit Cobb-Douglas MAUFs. This method is a sounder approach than traditional additive MAUFs since this type of utility function assumes neither a constant marginal rate of substitution between attributes nor total compensation between attributes, thus being more coherent with economic theory.

This new methodological approach has been empirically implemented to simulate farmers' behavior in a real case study, and the results obtained have been compared with those derived from two other well-known approaches, profit maximization and the additive MAUF. This study reaches two main conclusions. First, the approach proposed to elicit Cobb-Douglas MAUFs can be easily implemented in real settings, and therefore, it can be a useful procedure for ex-ante simulations of policy instruments or any type of future scenario. Second, this new method proposed based on the maximization of the Cobb-Douglass MAUF can produce fruitful outcomes for policy analysis because it provides better simulation results than more traditional approaches. Two pieces of evidence justify this assessment. First, calibration is more precise using this approach than in the other approaches compared since the resulting MAUF better reproduces farmers' current behavior. Second, the resulting demand curve has a smoother and more credible shape than those obtained from previous approaches since farmers are expected to make marginal changes when facing marginal external shocks.

However, it is also worth pointing out that the method proposed is based on some rather restrictive assumptions that can been seen as potential shortcomings. The strongest ones are: i) the assumption regarding utility-independence, allowing the MAUF became a separable function, and ii) the assumption about the stability of the MAUF, i.e. the parameters of the utility function do not change when circumstances do (farmers behave the same way whatever occurs). Because of both potential limitations, further research is required to confirm that this new non-interactive method to elicit Cobb-Douglas MAUFs represents a reasonable enough approximation to simulate real farmers' behavior. In this regard, some others functional forms of the utility function could be elicited and tested, such as the constant elasticity of substitution function (CES, a more general form than the Cobb-Douglas function) and other ones not assuming utility-independence. Moreover, it would be worthwhile implementing experiments to test that MAUF parameters remain constant over time (by using multiple elicitation procedures with the same decision-makers in different time periods).

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Capítulo 4
Water banks: What have we learnt from the international experience?

Water banks: What have we learnt from the international experience?[6]

Abstract

In recent decades, the use of economic instruments has been promoted as a way to improve water demand management, required due to the difficulty of further supply increase. Against this backdrop, this paper analyses the potential of water banks as a type of water market that can provide institutional flexibility in the allocation of water resources among different users. Research has involved an extensive review of the literature, which has allowed us to identify different types of water banks that operate around the world, as well as an analysis of the experiences of water banks implemented to date, in order to assess the performance of this economic instrument in improving water management. This has provided evidence that water banks, if properly implemented, can be a useful tool for improving governance of water resources. Finally, the analysis has enabled us to propose a number of guidelines on how to improve the implementation of water banks in different countries around the world.

Keywords: water policy, water management, economic instruments, water banks, water reallocation.

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4.1. Introducción

Population growth and the resulting demand for food (irrigation) has, over the course of the 20th century, led to a marked increase in global water abstraction and consumption (FAO, 2012). Moreover, as a result of global warming and climate change, we are witnessing an overall reduction in water availability around the world (structural scarcity), as well as more frequent and more severe drought periods (cyclical scarcity). This is especially true in arid and semi-arid regions (IPCC, 2014) such as California, Australia and Spain, where irrigated agriculture is particularly competitive, and agricultural consumption accounts for up to 80% of total water use.

To date, management strategies to cope with water scarcity have been predominantly based on structural solutions aiming at boosting the total amount of water supplied (increase in reservoir capacity, exploitation of groundwater or reusing treated wastewater). In mature water economies, however, evidence shows the difficulty of further implementation of this traditional approach to water policy based on supply-side measures (Randall, 1981), for both environmental reasons (almost all usable water resources are already used) and economic ones (the high costs of generating additional water supply). This situation has led to a reorientation in water policy, in order to make it more focused on the efficient reallocation of existing water resources. This is especially important in regions where it is not possible to further expand water supply ('closed' basins) and where the implementation of demand-side measures is thus considered a priority. Such measures include economic instruments such as water pricing, water caps, water markets and voluntary agreements. These mechanisms provide public authorities with useful tools for solving mismanagement problems through a more efficient reallocation of existing resources, thus mitigating the effect of both structural and cyclical scarcity (water shortages due to droughts) (Garrick *et al.*, 2011; Lago *et al.*, 2015).

Within this context, this study aims to examine the potential of water banks as an economic instrument for demand-side management in closed basins with environmental sustainability issues (overallocation of resources and deterioration of water bodies) and great uncertainty as to resource availability (growing impact of droughts due to climate change). Accordingly, this study includes a critical analysis of the implementation of water banks around the world in order to examine the advantages and disadvantages of this economic instrument for managing scarcity, environmental issues and uncertainty regarding water availability. It concludes by providing a number of suggestions for improving the design and implementation of this economic instrument in the sphere of water policy.

To that end, the paper is organized as follows. The following section is dedicated to defining the concept of water banks and identifying different categories. Section 3 describes the key experiences of water banks implemented around the world. Section 4 details the advantages and disadvantages of water banks as an instrument for managing scarcity. Lastly, in light of these experiences, Section 5 concludes by presenting a series of proposals for improving water banks as an instrument to ensure efficient use of resources across sectors and users, including the environment.

4.2. Water banks: concept and types

4.2.1. Water banks as a type of water market

Economic theory holds that markets are an efficient mechanism for allocating scarce resources in case several stringent conditions are met, including perfect competition, the absence of externalities and no transaction costs, among others (for further details, interested readers can consult any microeconomics handbook, such as Mas-Colell *et al*., 1995 or Gravelle and Rees, 2004). However, these conditions are not usually found in the real world, as is the case with water markets (Qureshi *et al*., 2009; Goemans and Pritchett, 2014). Nevertheless, water markets have been identified as an economic instrument that offers the potential to improve water management in contexts of scarcity, since water market reallocations can lead to more efficient water use and a significant improvement in social welfare (Hearne and Easter, 1997; Brooks and Harris, 2008; National Water Commission, 2011).

The term "water markets" actually refers to a whole range of institutions that facilitate voluntary exchanges of water between users. Indeed, these markets may take different forms depending on their defining variables, including key aspects such as their legal status (formal and informal), the rights being traded (permanent rights, temporary rights or 'spot markets', and options on temporary rights) or the parties involved (sellers and buyers). With respect to the latter, it is important to distinguish between water markets involving private parties, where buyers and sellers interact directly to negotiate the terms of water rights transfers with the possible involvement of intermediaries or professional brokers, and the so-called water banks, which operate in a more institutionalized context and which by definition involve agents who are not themselves water users. Thus, a water bank is a market mechanism through which an administrative agency (public or private) acts as an essential intermediary in the trading of rights.

This research paper focuses on the study of water banks, not to be confused with water banking, which is a resource management strategy based on water storage (Dellapenna, 2000). Indeed, the term 'water banking' refers to depositing water rights, either on paper or an actual volume of water, in a "bank", understood as the water stored in a reservoir, aquifer, or such like. This deposit in the bank provides its holder with access to a wide variety of operations, including deferred resource use and its transfer to other users. Water banking is a well-recognized policy tool to address similar challenges to water banks. Interested readers can consult the case of Arizona water banking as a successful example of this instrument (Megdal *et al.*, 2014).

Delacámara *et al*. (2015) define the concept of a water bank as an institutionalized and centralized process established to facilitate the transfer of water allocated to specific users or uses, to other users and uses. At its simplest, a water bank is a single intermediary acting between buyers and sellers of water rights, whether that transfer is temporary (spot) or permanent. Water banks are typically managed by a public institution (e.g., water agencies). In such cases, water is transferred from certain users to others under the supervision of the public administration, which verifies that the water transactions fulfil all legal requirements, sometimes including constraints linked to environmental and social criteria.

The underlying concept of the water bank is to serve as an institutional mechanism designed to respond to cyclical changes (by means of temporary transfers of water rights) and structural changes (by means of permanent transfers) in resource availability, reducing the transaction costs of operations. Water banks also help boost market activity and improve transparency by facilitating contact between buyers and sellers, as well as providing information on prices and quantities exchanged. This market system enables interested water users to lease or sell their water rights to any agent (whether user or non-user) willing to lease or purchase them, thus fostering a more efficient resource allocation in both the short and the long run. For all these reasons, this type of water market is becoming more prevalent in the more mature water economies of the world.

4.2.2. Water bank typology

In practice, the term 'water banks' can cover a wide variety of institutional designs that all adhere to the general concept of water banks described above. Accordingly, we examine below the diverse forms of this kind of water market, based on the experience of water banks implemented to date, and offer an analysis of their principal defining characteristics. This analysis of the characteristic variables of water banks has enabled us to identify a number of different types, which are described below.

First, it should be noted that water banks differ with respect to **the nature of the organization responsible for their implementation**. In this regard, we can observe the following types:

- *Public water banks* are organized and managed by a public administration, typically one with expertise in the field of water.
- *Private water banks* are organized and managed by means of a private initiative, generally run by non-profit organizations, such as NGOs dedicated to environmental conservation.

A second defining variable of water banks is the **type of rights being exchanged**. In this regard, we can distinguish between the following types:

- *Permanent water banks*. Rights-holders permanently transfer their water entitlements to the water bank. The rights acquired by the bank can subsequently be reassigned, partly or wholly, to other users (current rights-holders or new users), either by means of acquisition or via a system of free public concession. These banks can be aimed at solving problems associated with structural water scarcity, both economic and environmental in nature (Wheeler *et al.*, 2012; Rosegrant *et al.*, 2014; Hanak, 2015), as is discussed below.

- *Spot or temporary water banks*. These banks act in same way as permanent water banks but with the difference that they deal with temporary transfers of water use rights (usually for an irrigation season) or specific quantities of water (spot). In both cases, the activity is concentrated in periods of drought, and the aim is to mitigate the effects of cyclical shortages (Booker *et al.*, 2005; Kahil *et al.*, 2015).

- *Option contracts banks*. This type of water bank facilitates the exchange of contracts that provide buyers with the option (but not the obligation) to buy water from the seller (the holder of the water rights), in exchange for a certain price or "premium" (Jercich, 1997; Cui and Schreider, 2009). If the aforementioned option is eventually exercised, the buyer pays additional compensation to the seller, called the "strike price". These contracts allow the buyer to hedge against the risk of not having enough water for their activity, while simultaneously ensuring that the seller does not forfeit the water entitlement (the right to use the water) (Howitt, 1998; Rey *et al.*, 2016).

Water banks can also be categorized according to their **purpose**. In this regard, we can distinguish between:

- *Water banks for the reallocation of resources as a production input*. Exchanges of rights that enable water banks to reallocate the resource (temporarily or permanently) depending on current and potential suppliers and demanders according to market forces, fostering the transfer of water from lower-value to higher-value uses. These transfers, in the absence of negative externalities, help to make water use more economically efficient (Grafton *et al.*, 2012; Wheeler *et al.*, 2014).

- *Water banks for environmental purposes*. These banks work by purchasing rights without subsequently reallocating them (Wheeler *et al.*, 2013). This type of bank thus provides a solution to environmental problems stemming from both structural water shortages (addressing overallocation of basin resources by purchasing permanent rights) and cyclical shortages (tackling low water flows in the dry season by purchasing temporary rights).

- *Water banks for managing risk related to water availability*. Climate and hydrological uncertainty inherent to water management causes interannual variability of resource provision. This exposes users to significant risk, and as a result they do not make economically-efficient decisions (Alcón *et al.*, 2014). In order to minimize sub-optimal decisions and improve water-use efficiency, these banks work by negotiating water options contracts. This helps to improve supply security for the buyers of water options contracts (by reducing supply security for the sellers of such contracts), thereby enabling an effective transfer of risk between users with different levels of risk aversion (Howitt, 1998; Rey *et al.*, 2016).

Lastly, focusing on **management strategy** allows us to differentiate between:

- *Active water banks*. Those where the managers of the bank adopt a proactive strategy as "market makers", buying water rights out of the bank’s own budget, and subsequently attempting to sell them to potentially interested users. In this regard, the aim of the water bank management is to achieve a balanced market, by trying to ensure that the sum of purchases and sales does not yield a net cost (the amount spent on purchases should equal revenue from sales), or that said cost does not exceed a maximum budgeted for this purpose. It should be noted that in these cases the bank administrator is the one who sets the conditions for the purchase and sale of rights (or options), and these banks thus become a type of monopolistic market with a one-way trading system (Loomis *et al.*, 2003). As such, the bank first acts as the sole buyer of water rights or options (monopsony through public purchase tenders), and then in turn becomes the sole vendor of such rights or options (monopoly through public sale offerings).

The purchasing system can vary depending on the nature of the public offerings. The bank may: i) establish the maximum amount of net purchases (maximum spending budget), whether by means of a fixed price or via auction (successive increments in the purchase price until the total allocated budget has been spent); ii) fix a maximum volume of water to be acquired, also by means of either budget limits or auction; or iii) establish a fixed market price for acquisitions, without budgetary constraints or limits on the volumes of water to be purchased. These public offerings may also be differentiated according to whether they are open or restricted; whereas all rights-holders in the territory under the bank's jurisdiction (e.g., a river basin district or users of an aquifer) can voluntarily attend the former, the latter is only for certain types of specifically-authorized users.

Similarly, public sale offerings can be differentiated in terms of both price conditions and contract amount, as well as with respect to their open/restricted nature. Active water banks are a useful way of boosting market activity (improving economic efficiency) and exercising more effective control over market operations (reducing externalities and minimizing asymmetrical information about water prices).

- *Passive water banks* limit themselves to facilitating contact between buyers and sellers so that operations can be carried out according to the supply and demand at any particular moment. In these cases, the role of the manager of the bank is simply to act as an intermediary for purchases and sales (broker), either as a clearinghouse or through sealed bid double auctions. In thinner water markets, water banks usually adopt the first approach (clearinghouses), where buyers and sellers reveal their intent to buy and sell, usually posted on bulletin boards, and trades are executed when matching offers and bids are found in terms of quantity and price (Hadjigeorgalis, 2009). Sealed bid double-auctions only take place in more liquid water markets. The offers to buy and sell rights are based on a system similar to the stock market, where the bank provides up-to-date and transparent information (positions or offers to buy and sell). Thus, by matching existing purchase and sales offers, a market clearing price is achieved—the price at which the exchange of all the rights of all those wishing to accept/pay the equilibrium price are cleared (Bjornlund, 2003).

Having analysed the different designs of water banks, and set out a typology based on the existing range of features, it should be noted that not all combinations of defining variables can be found in reality. Indeed, the experiences of banks implemented to date reveal a strong correlation between some of the variables analysed above. For example, it has been observed that banks created to tackle environmental problems arising from the overallocation of rights, tend to be active banks dealing in permanent rights. Alternatively, when it comes to minimizing the economic consequences resulting from shortages caused by periodic drought, the water banks set up are public and exchange temporary water rights, either through active or passive management strategies.

4.3. International experiences

To date, there have been many experiences of water banks developed around the world. Of particular note are those from the western states of the US (particularly California), the southern states of Australia, and from Spain, where the success of this economic instrument in improving drought management has been clearly demonstrated. It is no coincidence that the most prominent examples of water banks have been developed in these areas, given the similarities between all these territories in terms of climate (Mediterranean climates with high variability in resource availability), hydrology (high water demand and closed basins) and production (highly profitable agricultural uses competing with urban and industrial uses).

The categorization detailed above has enabled us to frame the experiences from around the world according to the different types identified. Before analysing them in detail, a summary of the types of water banks implemented in each territory is shown in Table 4.1.

Table 4.1. *Water bank experiences*

	Type	*California (Federal and State)*	*California (Federal)*	*Idaho (State)*	*Montana (NGO)*	*Colorado (State)*	*Colorado (NGO)*	*New Mexico (State)*	*New Mexico (Private)*	*Texas (State)*	*Texas (NGO)*	*Oregon (State and NGO)*	*Washington (State)*	*Washington (NGO)*	*Australia (Private)*	*Australia (State)*	*Spain (State)*	*Chile* (State)*
Rights exchanged	Permanent			X				X		X	X		X	X	X	X	X	
	Temporary or *spot*	X		X	X	X	X	X	X	X	X	X	X	X	X		X	X
	Options		X											X				
Purpose	Resource reallocation	X				X			X	X			X		X		X	X
	Environmental	X		X	X		X	X			X	X	X	X		X	X	
	Risk management		X															
Management strategy	Active	X	X	X	X	X	X	X			X	X	X	X		X	X	
	Passive					X		X	X	X					X			X

* Water banks experience in this country is limited to a pilot project.

4.3.1. Water banks in California

The recurrent drought episodes that have hit this American state are key to understanding the evolution of its water management policy. In this context, the severe drought that hit the state between 1920 and 1930, combined with a particular set of circumstances including the availability of large federal subsidies, created a political opportunity to develop numerous large-scale water projects in the early to mid-twentieth century. These infrastructures combined to create an extensive hydraulic network connecting all its counties. For decades, this network allowed those users with greatest needs in terms of supply security to establish water storage and conveyance infrastructure in the north of the state (where it is wetter) and transfer resources, when necessary, to the south of the state (drier) (Griffin, 2016). From the 1970s onwards, the mature phase of the Californian water economy was reflected in its inability to increase supply. That marked the beginning of the change in water policies, with a focus on new instruments for reallocating existing rights, and the origin of the water markets. The severe drought of 1976-1977 was a crucial moment, driving the Bureau of Reclamation to introduce the first Californian water bank to facilitate water trading between public water entities, especially for urban supply. Over the course of that year, the bank bought 57 Mm^3 (1 Mm^3 = 1 hm^3 = 1 GL = 810.7 acre-feet), of which 52 Mm^3 was subsequently resold (Lund *et al.*, 1992). Nevertheless, the main problem facing this first bank was the restriction imposed on private users, which prevented farmers—the main users of water in the state—from accessing the resource (Howe, 1997).

The next major milestone was brought about by the great drought that began in California in 1987. Among the measures implemented in response to this challenge, particularly notable was the Californian state government's creation of the Drought Emergency Water Bank in 1991. This water bank was designed so that the state could act as an agent with an active management strategy, aiming to facilitate temporary transfers of water from the agricultural sector to urban use, at a price set by the state government (Lund *et al.*, 1992). The bank's purchase transactions were conducted through various types of one-year contracts (Jercich, 1997; Clifford *et al.*, 2004): in the first type of contract, irrigators sold surface water and agreed to stop cultivating crops; in the second type of contract, farmers sold surface water so that the bank could use it at any point along its watercourse, but they could continue irrigation by pumping groundwater; and in the third type of contract, the bank had access to the reserves held by the seller. This water bank was more successful than anyone had anticipated, producing the largest number of regional exchanges of water resources that had ever taken place in the US up until that point. In total, 1,012 Mm^3 of water was acquired, of which 50% came from 348 fallowing contracts (the first type of contract), 32% came from 19 of the second type of contract, and the remaining 18% was purchased via the third type of contract, through which 181 Mm^3 of water was acquired with only 4 contracts (Israel and Lund, 1995). The water bank subsequently reallocated a total of 488 Mm^3 among higher-value uses; this was water that had previously been purchased at a price of \$0.10/m^3 before being sold at a price of \$0.14/m^3, thereby mitigating the economic losses caused by the drought (Howitt, 1998). Of the total reallocations, 30% (150 Mm^3) was assigned to environmental purposes, in order to increase flows in the Sacramento-San Joaquin delta, and the rest was left unsold (Howitt, 1994).

The operations of this water bank were extended in 1992 but with a much lower intensity of market activity, with volumes of water reallocated dropping to 235 Mm^3 (Hadjigeorgalis, 2009). This decrease in the water bank's activity, both in terms of the quantity of water transferred as well as with respect to number of participants, was triggered by the heavy rains that fell in early 1992, which eased the shortage situation and reduced demand on the water bank. Despite the rains of 1992, California's hydrological situation did not return to normal, and in 1994 a new water bank was set up, which bought 272 Mm^3. The forecast of an extremely dry year prompted the State Department of Water Resources to set up an options bank in late 1994. Through this bank, option rights were purchased for a total of 36 Mm^3, at a premium of $0.003/$m^3$, and exercised price in the range of $ 0.030-0.035/m^3 (Howitt, 1998). Finally, due to the heavy rains that fell in 1995, which finally brought the drought of previous years to an end, these options contracts were not exercised. In any case, this experience showed that advance planning can be an efficient way to tackle potential droughts, helping to improve water supply security for users with higher water productivity and greater risk aversion (Jercich, 1997).

In 2009, attempts were made to set up a new water bank in California to tackle a new drought. This time, however, the experience was unsuccessful because: i) the institutional instrument design was extremely complex, meaning that only a few operations could be carried out, and ii) exchanges from water exporting areas were blocked as a result of protests by environmental organizations (Medellín-Azuara *et al.*, 2013). This failure highlighted the need to design water banks taking into account not only potential market participants (current water users) but also other stakeholders (civil society organizations) in order to ensure that the bank's exchanges of the resource are considered universally beneficial for all parties involved.

Furthermore, it is worth commenting that although 2014 and 2015 were the two hottest years in the state's recorded history and caused another severe water deficit, no water banks were implemented as an instrument to improve resilience to drought (Hanak, 2015).

To sum up the Californian experience of water banks, it can be described as generally positive, in that this type of market has helped minimize the economic and environmental impacts of droughts. Nevertheless, some authors (Hanak, 2003; Howitt, 2014) have also highlighted negative aspects, such as greater overexploitation of groundwater and a decline in economic activity in water-origin areas. These negative effects have become a matter of current social concern that has led to water banks being ruled out as a way of managing the more recent droughts.

4.3.2. Water banks in other western US states

In the western US states there are a wide variety of regions that, like California, are semi-arid and prone to highly variable rainfall and water regimes. This climate variability causes droughts and, in an effort to reduce the consequent adverse effects, many of these states have implemented water banks, among other measures. Below is a brief summary of relevant examples of water banks developed in these states.

Idaho was one of the first states to begin to set up water banks. This state has a long tradition of implementing water banking activities, allowing agents to not only store unused water in pools (sets of water rights "deposited" and physically stored for water banking activities.) for future use, but also to sell stored water to third parties. However, it was not until 1995 that the state government formally developed a water bank that enacted active strategies to acquire, for environmental purposes, temporary and permanent water rights from the pools of the Upper Snake, Payette and Boise River Basins. Subsequently, in 2001, the Lemhi River Basin water bank was created with the same purpose (WestWater Research, 2003). The objective of these water banks is to correct environmental damage caused by the federal government's large hydroelectric dams in the Columbia basin (with a special focus on salmon recovery), for which they receive funds from the federal agency, Bonneville Power Administration (BPA) (Clifford *et al.*, 2004).

Montana is in the Columbia River Basin, and so this state also benefits from the funds channelled through the BPA. However, in this state, those funds are managed by an environmental NGO. The NGO uses the funds to buy rights through a water bank which, in this case, is referred to as a water trust. The Montana Water Trust, in operation since 2001, aims to restore the water flows in the Columbia River Basin.

A water bank was created in **Colorado** in 2001 to encourage the reallocation of resources among users of the Arkansas River Basin. This state-run bank initiative was designed to implement a passive strategy via an online platform, with its operations limited to facilitating contact between buyers and sellers, and intervening as a clearinghouse for the operations, which were closed electronically. However, this attempt to institutionalize a water bank failed and not a single transaction was completed. A fundamental design error was the publication of the details of users interested in making a transaction, which allowed sellers and buyers to contact each other directly without the intermediation of the bank, thus reducing their transaction costs of operations. Following this unsatisfactory experience, the state of Colorado created another water bank in 2009, which this time was successful: the Colorado West Slope Bank. It is an active bank that was established for the reallocation of resources from long-standing rights holders (seniors), to more recent users (juniors). Lastly, attention should be drawn to the Colorado Water Trust, whose aim is to purchase temporary water rights for environmental protection purposes.

In the case of **New Mexico**, different types of water banks have been set up: both state-run and private initiatives, and implementing either active or passive strategies. The state water banks were created to acquire rights, both temporary and permanent, as a way of preserving the flow of the Pecos River. All of them, however, registered limited or no activity, with the exception of the Pecos River Water Lease/Purchase Program, which generated moderate levels of activity to fulfil the requirements of water flows around the Texas state-line. Private banks, on the other hand, are oriented exclusively towards resource reallocation, having been developed by their organizers as for-profit entities (O'Donnell and Colby, 2010).

Another state where a water bank has been set up is **Texas**, with the creation of the Texas Water Bank in 1993 as a mechanism to allow voluntary transfers of water rights between sellers and buyers, either temporarily or permanently. The bank acts as a clearinghouse of sales transactions and keeps a record of the activity carried out. However, the activity of the Texas Water Bank has been very limited due to poor institutional design resulting in high institutional transaction costs, limited public awareness, inadequate rules for groundwater operations, and issues with water rights cancellation statutes, among other problems. Indeed, one particular obstacle was the prior existence of efficient brokers that competed with the bank by operating with lower transactions costs (Clifford *et al.*, 2004). In addition, the Texas Water Trust was established in 1997 and is responsible for sourcing donations from individuals, companies and institutions for the lease or permanent purchase of water rights for environmental purposes.

There have been water banks in **Oregon** since 1993, the year in which the Oregon Water Trust was established, becoming the first organisation of its kind in the world. It is an environmental protection organization also dedicated to restoring the flows of the state’s rivers.

As a final example from the US, the case of **Washington State** should be mentioned. The not-for-profit Washington Water Trust has, since 1998, been responsible for restoring natural flows of water in the Yakima and Dungeness Rivers. It does so by means of public offers to buy temporary rights (in drought years) and permanent water rights (Cronin, 2015), in addition to contracts for long-term options. In the same state, the Dungeness Water Exchange has been in operation since 2013, created by the Washington State Department of Ecology to run two demand-management programmes. The first, called the ‘mitigation programme’, consists of the establishment of a water bank so that new domestic users have access to water. In the absence of greater water availability, these users must purchase a certificate that guarantees that they will fulfil any of the water conservation options established for this purpose. The money raised by the certificates will go to purchase water from willing sellers. However, it is worth commenting that this certificate has proved to be something of an obstacle because water users cannot fully participate in the market until the water rights certification process is completed, and this process can take a long time to complete. The second programme, called the ‘restoration programme’, uses state, federal and private money to purchase water in order to restore the river's flow.

4.3.3. Water Banks in Australia

Australia has the most active water markets in the world. This market has been in operation since it was first approved in the 1980s, initially only for temporary rights transactions, but as of the 1990s permanent rights have also been traded. In fact, it is estimated that approximately 20% of the water used in this country comes from commercial transactions. In this context, it is worth mentioning that a great deal of the water transactions in Australia are carried out with the support of intermediaries, including brokers and lawyers. These intermediaries have gained the confidence of water users –predominantly irrigators–, and some have subsequently gone on to create and manage water banks operating as clearinghouses that are intensively used for water trading. In fact, as Bjornlund and McKay (2001) point out, local brokers are preferred for arranging transfers locally, but water banks are favoured for long-distance transfers. The aim of these private water banks is to promote resource reallocation driven by market forces, adapting supply and demand in both the short (temporary rights transfers) and long term (permanent rights transfers). Exchanges of water in these banks are performed via two passive-strategy mechanisms: i) through the internet, by means of a bulletin board where buyers and sellers publicize their offerings, allowing operations to be matched up in terms of price and volume; and ii) through sealed bid double auction, as they are in the stock markets (Bjornlund, 2003). The volume of operations and adaptation of market prices to changing conditions suggest that these water banks have been successful as a reallocation instrument for available water resources.

From 2004, as a result of the severe impact of the so-called "Millennium drought" on the status of its water bodies, successive programmes have been proposed in Australia for purchasing permanent rights for environmental purposes (mainly through the 'Living Murray Initiative' and 'Restoring the Balance' programmes). These programmes have acted as public water banks and make public offers to purchase rights (active strategy) charged to the public budget (Wheeler *et al.*, 2012).

4.3.4. Water banks in Spain

Water markets and water banks were introduced in Spain in 1999, when the law in force at that time was amended. The reform set out the regulations for the creation and operation of water exchange centers. According to Spanish law, water banks in Spain can only be publicly-run and require authorization by the Cabinet. In addition, these exchange centers, which adopt an active strategy, act only in "exceptional situations of water scarcity" (special drought situations or severe overexploitation of aquifers). Once set up, these centers operate through Public Offers of Acquisition of Rights (OPAD in Spanish), whether temporary or permanent, in order to reallocate water among users in need of the resource or to improve the status of water bodies.

There has been a relatively low level of activity in the Spanish water markets, and it has only occurred in periods of drought. In fact, in the market's busiest year (2007) the volume of water exchanged amounted to less than 0.5% of the total water used at national level (Palomo-Hierro *et al.*, 2015). In any case, most water exchanges have been performed through lease contracts; only a quarter of the operations have been conducted through water banks.

To date, three banks have been created in Spain that have acted in exceptional water situations. These exchange centers are in the south-eastern part of the country, which is the driest area with a similar climate to the regions discussed above. Below is a brief summary of these water banks.

The first Spanish water bank was set up in the Guadiana River Basin, in response to the overexploitation of the Mancha Occidental aquifer; the existing water rights concessions represent more than twice the amount of the available resource. Between 2006 and 2008, six offers for permanent purchase were made as a way to restore the water balance of the aquifer. The prices paid related to units of irrigated area (on completion of the purchase sellers would have to stop irrigation) rather than units of water volume. Prices varied depending on the type of crop (annual or perennial) and proximity to the most deteriorated area. If we translate those prices into volumes of water associated with the surface for which water rights were acquired, prices ranged between €2.28 and €2.35/m^3, with a total cost of just over €65 million. There is no firm consensus as to the volume of water the bank was able to buy back, because in some cases water that had not been used in recent years was bought (sleeper rights), and also some of the acquired rights were allocated as grants for social crops such as grapevines (WWF España, 2012). The activity of this water bank, which was expected to continue over time, was halted as a result of the economic and budgetary crisis that Spain has been suffering since 2009, which forced cuts to public spending on environmental policies such as these.

The second water bank experience occurred during the drought of 2005 to 2008 in the Júcar River Basin, where temporary water rights were purchased to maintain water flows in the middle stretch of the Júcar River, between 2006 and 2008. For the four offers made, a fixed price was set based on the water apparent productivity of the crops in the area. As a result, the initiative resulted in the acquisition of a total water volume of 77.9 Mm^3 at an average price of €0.19/m^3 in 2006 and €0.25/m^3 in 2007 and 2008. The experience was considered fairly successful because substantial improvement was reported in the flow in the middle stretch of the Júcar River (Garrido *et al.*, 2013).

During the same drought, a water bank was established in the Segura River Basin and was in operation between 2007 and 2008. This water bank also made offers to purchase temporary water rights, with the aim of restoring environmental flows in the Mundo and Segura Rivers and ensuring water supply to local populations. These offers were successful in buying back about 6 Mm^3 at an average price of €0.17/m^3, which was eventually allocated entirely to the improvement of the environmental flows of the rivers (Garrido *et al.*, 2012).

4.3.5. Water Banks in Chile

The water market in Chile is known for its free-market doctrine (Bauer, 1997), with fully decentralized and deregulated operations. This institutional design for the development of water markets has allowed the establishment of an active market for water rights among users, which has enabled both cyclical and structural shortages to be tackled. However, a number of criticisms have been levelled at its performance: imperfect market structure (oligopolies), lack of information on the operations carried out and poorly-defined water use rights (Hearne and Donoso, 2005). These factors have caused high volatility in prices. Despite the great concern that exists in this country on how to improve the management of these markets, to date there has been no institutional reform in this regard. In any case, it is worth outlining the results of the pilot project developed in the Limarí River as part of the "Electronic Market for Water Rights" (MEDA in Spanish) initiative. This initiative consisted of implementing a broker-style online trading platform with the aim of promoting transactions between potential buyers and sellers. This pilot project was in operation for several years, but ceased operations due to lack of public funding (MEDA ended in April 2011).

4.4. Water banks as an instrument for managing water scarcity

Despite the potential displayed by water markets and water banks as instruments for efficient management of water scarcity (both structural and cyclical), a study of the experiences of water banks around the world reveals how their operation entails a number of advantages and disadvantages, which should be taken into account to ensure the appropriate design and implementation of these economic instruments. In this regard, an extensive review of the existing literature on the subject has been carried out in order to properly catalogue the main advantages and disadvantages associated with water banks, as summarized below.

4.4.1. Advantages of water banks

Since water banks are a type of water market, they offer the same advantages, mainly related to improved efficiency in the use of water (Israel and Lund, 1995; Easter *et al.*, 1999; Garrick *et al.*, 2009; Grafton *et al.*, 2011; Rosegrant *et al.*, 2014):

- They increase utility (income in the case of private, profit-maximizing agents) for all market agents (buyers and sellers of water). The participation of water users in the market is always voluntary, which ensures that all operations are beneficial (raised utility or income) to both parties.

- They improve resource-allocation efficiency, encouraging water transfers from activities of lower value of marginal utility (value of marginal productivity in the case of productive economic uses) towards activities with higher value of marginal utility, thereby maximizing the total utility (production value in the case of productive economic uses) generated by all agents participating in the market. As a result, in those cases where externalities are minimized, water banks usually lead to improved social welfare (net benefits from a public perspective).

- Market prices provide a proxy of the true opportunity cost of water, encouraging more rational use of the resource.

- They can ensure better supply security to the users that are most averse to the risk of hydrological uncertainty, since they provide the possibility of water exchange in times of water shortage.

- They rationalize the construction of new infrastructure projects aimed at increasing water supply, as the markets provide an alternative to building expensive water works (when market prices are lower than the marginal cost of new resources).

In addition to the advantages they share with other water markets, water banks offer a number of specific advantages (Bjornlund and McKay, 2002; Clifford *et al.*, 2004; O'Donnell and Colby, 2010; Garrick *et al.*, 2013b; Gómez-Ramos, 2013; Rey *et al.*, 2014):

- Water banks centralize purchases and sales of water rights (or options), reducing operational or static transaction costs for both the agents involved in the market and the institution managing the bank. Transaction costs are a key issue in environmental policy (Garrick *et al.*, 2013a), and are also a key factor determining the performance of water institutions (e.g., water banks) (Challen, 2000). For a recent review of transaction costs in environmental policy, interested readers can consult McCann *et al.* (2005) and Marshall (2013). With a more specific focus on water markets, Garrick *et al.*, (2013b) and McCann and Garrick (2014) are also worth referring to. Following recent debate, transaction costs can be divided into two categories. The first refers to the costs of designing and setting up the instrument under analysis (namely institutional transaction costs) while the second are associated with the operational costs of the instrument (namely static transaction costs). These static costs include: i) support and administration costs; ii) contracting costs; iii) monitoring and detection costs; and iv) prosecution and enforcement costs. Water banks are particularly effective at reducing contracting costs. This advantage is the most important one for market agents, since it involves the costs of finding parties interested in participating, bargaining costs and decision costs associated with transactions. The reduction of all these static transaction costs boosts water trade by making operations more profitable (or ensuring they generate more utility in the case of public or non-profit organizations) for buyers and sellers.

- Water banks encourage governmental oversight of environmental and social externalities arising from water transactions. They also allow operations with environmental purposes (public offers to purchase rights without subsequent reallocation), in order to increase river flows, restore overexploited groundwater bodies, etc.
- They make the market more transparent by releasing purchase/sale prices and making them publicly available to all users.
- Public initiative water banks, since they are managed by the government, provide greater security and reassure buyers as to the actual availability of negotiated water resources.
- The implementation of water banks during the early stages of drought periods should more effectively raise all users' awareness of the need for efforts to reduce demand in order to mitigate the negative effects of drought.

4.4.2. Disadvantages of water banks

There are a number of disadvantages associated with water markets, and which are therefore also associated with water banks. Below is a list of the main drawbacks inherent in the use of these economic instruments (Dinar *et al.*, 1997; Hearne and Easter, 1997; Bjornlund and McKay, 2002; Bjornlund *et al.*, 2007; Qureshi *et al.*, 2009; Garrido *et al.*, 2013):

- They could generate negative environmental externalities, especially by altering water flows in natural watercourses. This occurs firstly through changes in the location of uses, which can reduce flows (sale of water from the lower to the upper part of the basin) or increase flows (sales in the opposite direction). Secondly, it occurs through the overall decline in returns when transfers are made from areas of low-efficiency water use to areas of higher-efficiency water use, resulting in an increase in water depletion (reduction of natural flows) at basin level.

- They could generate social externalities in the areas of origin, due to the loss of employment caused by abandoning productive activity, which can in turn cause depopulation and territorial imbalances. Such issues could then present a political problem (e.g., rural stakeholders in areas of origin lobbying to maintain irrigated agriculture).
- The activation of "sleeper rights" or "paper rights". The presence of the market encourages the activation of these rights, resulting in an increase in the water abstraction from the system, a situation that exacerbates water shortages.
- Existence of other market imperfections, resulting from the small number of buyers and/or sellers, the variety of water rights exchanged and/or the lack of transparency in the information on volumes transferred and prices negotiated. A further consequence of all this is that the balance achieved by the markets is sub-optimal from an economic efficiency perspective. In this regard, it is worth noting the impact on the market of cultural barriers (unwillingness to use water markets as water is not considered a tradable commodity), physical barriers (lack of appropriate infrastructure for completing transactions) and legal barriers, which limit the number of agents that can operate in it.

It should be noted that water banks, as well as sharing the same characteristics as water markets, also have some specific features which result in a number of unique disadvantages not displayed by other water markets. In this regard, it is worth highlighting the complexity of the institutional design required for the creation and operation of water banks, especially in contexts where there is little prior experience with water markets (Embid, 2013). Consequently, water banks have greater institutional transaction costs, understood as the costs of designing and setting up this instrument incurred by the institution responsible for its creation (Marshall, 2013). This is because the establishment of a water bank requires high levels of investment and administrative management capacity. In addition, if it is an active water bank, it requires a large budget to carry out acquisitions and to be able to withstand the risk associated with such operations (possibility of losses). For these reasons, the organization and implementation of this instrument is generally only feasible for public administrations.

To summarize, we believe that the advantages of water banks outweigh the disadvantages, many of which can be limited through appropriate institutional design of the instrument. As a result, this type of market is becoming ever more common around the world as an economic instrument for improving efficiency of water use and mitigating the effects of water shortages (cyclical or structural) (Ghosh *et al.*, 2014; Rosegrant *et al.*, 2014).

4.4.3. Critical features: Defining successful water banks

The decision-makers responsible for water policy face the challenge of designing economic instruments for the improvement of water management (OECD, 2015). Accordingly, analysing related international experiences can be extremely informative, revealing the key factors behind the success or failure of such initiatives. In the analysis of water bank experiences, three key elements should be taken into account as determinants of performance: i) the water economy context, ii) the institutional context, and iii) the social context.

In terms of the *water economy context*, the development of successful water banks (or any other market-based instrument) requires a "mature water economy" (Randall, 1981), that is, an economy with a high but still growing demand for water combined with an inelastic water supply in the long run due to the limited possibilities of securing new water resources. Within this framework, policy reforms should involve two linked components: cap (imposition of diversion/extraction limits to avoid further sustainability problems) and trade (establishing tradable water rights to enable more flexible water reallocation) (Garrick *et al.*, 2013b). Thus, the use of markets mechanisms has been encouraged to be (cautiously) implemented in mature water economies (European Commission, 2012; Llop and Ponce-Alifonso, 2016). In this sense, water banks can be useful in two different ways. First, by directing permanent transfers of water rights towards higher-value uses, and second, by allowing temporary water transfers in order to manage periods of water scarcity (droughts). Considering the widespread and long-run impacts of permanent water transfers, some doubts arise as to the suitability of instruments that rely purely on private incentives (market-based instruments such as water banks) to reallocate permanent water rights. On the contrary, permanent transfers dictated by the public interest (i.e., public acquisitions aimed at improving social welfare) can be justified both in terms of reallocation between economic users and for environmental purposes.

In any case, temporary water banks have been proven to be a useful instrument for managing cyclical water shortages (droughts), providing a flexible and cost-effective tool for reallocating water from lower-value and more drought-resilient uses (e.g., irrigation farmers with herbaceous crops) to higher-value uses, both for private operators (e.g., urban water suppliers) and for environmental purposes (e.g., maintenance of water flows by public authorities or environmental NGOs).

Within the *institutional context*, it should first be noted that water rights must be clearly defined before implementing water banks. In fact, the creation of a centralized register of water entitlements exactly defining water allocations, use permits, etc. it is a strictly necessary condition for an adequate performance of any market mechanisms (Young, 2014a; Santato *et al.*, 2016). Particulary important is water use priority, which can vary greatly among users and sectors and especially where prior appropriation rules exist, meaning that water users in the same basin can have different water rights. This makes it difficult for these heterogeneous rights to be traded thought permanent water banks. In this regard, an interesting feature of temporary water banks is that they do not require homogeneity of water rights, since only water volumes (not rights) are traded; this, along with the doubts about permanent transfers driven by private incentives, means that temporary water banks are more likely to succeed than permanent ones. In fact, successful experiences of permanent water banks only can be found in cases where they have been managed by a public authority (public control of long-run externalities) and/or for environmental purposes (a public administration or NGO purchasing rights to reduce water depletion at basin level).

Moreover, it should be noted that temporary water banks for the reallocation of water rights as a production input have had notable success in basins where spot water markets had already been implemented. This evidence suggests that spot water markets can be considered as a first stage in the implementation of market-based instruments in mature water economies. Once the private agents involved have confirmed the potential profitability of making water transfers via this decentralized instrument, market activity can be expanded (and economic efficiency improved) in a second stage by the implementation of temporary water banks. The success of this approach to the design of water banks is supported by two key characteristics of the process. First, the prior implementation of spot water markets generates the required 'market training' for private operators, showing the economic value of water resources (market prices) and demonstrating how water transfers can be profitable for both buyers and sellers. Second, the activity of these market-based instruments usually occurs in a thin market at first (water users are reluctant to sell water for cultural reasons), and so at this initial stage it is only worth implementing simple institutional arrangements such as spot water markets. Only when market operations reach a critical mass are the investments needed to create a water bank (institutional transaction costs) justified, regardless of whether it is a public or a private initiative. With large enough markets, water banks become a more suitable instrument than water markets since they reduce static transaction costs in relative terms, thus leading to more efficient market performance (more operations and more value creation).

In addition to the accurate definition of water rights and previous experience with spot water markets, an appropriate design for water banks is also a key determining factor in the success of this instrument. We can cite a few examples of poorly designed water banks in order to illustrate this point. For instance, the Arkansas River Basin Water Bank Pilot Program failed mainly because it was designed as a passive water bank that published detailed information about all willing buyers and sellers. This made it possible for interested agents to arrange sales agreements outside the bank by contacting each other directly, thus avoiding the bank fees. Another clear example of failed design is the Texas Water Bank, which has registered minimal activity since 1993. This is, among other reasons, because it was designed as a complex operational institution (high institutional and static transaction costs) simply acting as a clearinghouse in a region with a long tradition of spot water markets with efficient private brokers that, thanks to their lower costs, were able to successfully compete with the bank for market operations. Lastly, it is also worth highlighting the case of the Guadiana River Basin Exchange Centre (Spain), an active water bank created for environmental purposes, which partially failed in its objective because the public offers to acquire rights had the effect of activating rights that had not actually been used in recent years (sleeper rights). In this case (and with other similar banks implemented in Australia), the failure to correctly define which rights were eligible for purchase in the public offers resulted in large public expenditure without any environmental improvement (real reduction in water abstractions).

Finally, regarding the social context, it should be pointed out that water banks have undergone remarkable development in areas where environmental problems (e.g., extractions over sustainable limits) are a source of social concern. In an attempt to minimize these problems, water banks have proven to be a useful tool for balancing water extractions, by leasing (temporary) and purchasing (permanently) water rights without subsequent reallocation. In these cases, water banks have allowed public administrations and environmental NGOs (e.g., water trusts) to participate in the market in order to reallocate water for environmental purposes, with these operations being covered either by the public budget (the society as a whole) or by voluntary private support (voluntary contributions from individuals and private institutions), respectively.

Furthermore, in order to ensure a successful implementation of water banks for resource allocation, any social concerns about negative externalities (environmental or social) should also be taken into account. The Californian experience is quite instructive in this regard: despite the successful experience in 1991 of using water banks as an instrument to manage droughts, this instrument failed when implemented in 2009. One of the reasons behind this failure was the opposition from particular sectors of Californian civil society which noted that water transfers would not be beneficial to society as a whole due to negative environmental and social effects. This case illustrates that in order to achieve a successful development and operation of water banks, it is not only the current water users that must be taken into account but also other stakeholders, in order to ensure as far as possible that water exchanges are perceived as beneficial for the whole society.

4.5. Conclusions

The analysis conducted reveals that water banks in the western United States and Australia are active and well-established. In other countries such as Spain, where such water markets have been in place for less than a decade, operations are gradually becoming more established (Palomo-Hierro *et al.*, 2015). The analysis of these international experiences with water banks has demonstrated that water banks are a market mechanism that facilitates transfers of water towards uses of greater value, including environmental uses. Thus, it can be said that this economic instrument is a useful tool for minimizing the negative impacts of water scarcity, whether cyclical or permanent. Indeed, the introduction of this type of water market makes the allocation of water use more efficient and also provides a tool to (partially) solve environmental problems linked to the overexploitation of water bodies. Moreover, compared with other types of water markets, water banks reduce the static transaction costs associated with support and administration, contracting, monitoring and detection, and prosecution and enforcement, thus helping to create more active water markets (increased economic efficiency). They also centralize market operations, which allows the administration (or other agency) to properly control potential negative externalities and prevent any kind of harmful speculation.

Nonetheless, the review carried out has also exposed some shortcomings in the implementation of water banks around the world. In this regard, a number of suggestions for improvement that the authors believe would help minimize or overcome these drawbacks are presented.

As with any water market, water banks can generate two main kinds of environmental externalities. The first is the change in water flow regimes, which could be harmful for aquatic ecosystems. In order to address this potential externality, operational rules of water banks need to include criteria for approving transfers, which guarantee that they are compatible with maintaining minimum environmental flows in all natural water courses affected. The other environmental externality is the likely increase in overall water consumption at basin level because more efficient irrigation practices lead to less water flowed back to water bodies (Young, 2014b). In order to reduce this negative environmental externality, operating rules of water banks must ensure that the total water rights transferred tally with the volume of water actually consumed (water extracted from the source that does not return to water bodies) in previous years. This is the only way of verifying that water bank operations do not increase water depletion, which would increase the quantitative pressure on water bodies and reduce water flows in natural watercourses. In this regard, a two-step approach is proposed. First, appropriate mechanisms are required to prevent the transfer of water rights that are not being used (sleeper rights), a situation that leads to an increase in total abstractions. Second, rules must be put in place to limit the amount of water transferred to the amount actually 'consumed' by the rights-holders (only the water evapotranspired by crops in case of irrigators), rather than the amount 'used' (total water abstracted from the source). In other words, water banks should avoid transferring the fraction of water corresponding to return flows. This is the only way to make sure that the water bodies in the areas of origin maintain the levels of water extraction that predate the banks' operations (Delacámara *et al.*, 2015). To that end, both the volume of water effectively used in previous years and the technical efficiency in water use would have to be determined in order to calculate the volume of return flows. Consequently, the volume of transferable water should be limited to the water abstracted from the water bodies minus the returns that would have originally occurred.

Transaction costs are also a relevant issue when designing and implementing water banks. As McCann (2013) shows, a monopsony structure as the one provided by water banks may facilitate bargaining, easing contact between users and a central operator instead of between users, thus reducing static transaction costs. In any case, an efficient initial design of these banks is required in order to minimize institutional transaction costs. Before creating this kind of centralized market, bank developers should be encouraged to examine the current water market framework: agents, 'market training' (experience in water markets), number of agents previously operating in the market and volume of water or rights traded, the role of private brokers, etc. This analysis should help inform decisions regarding the timing (when to create the water bank) and structure (market infrastructures and design of operational rules) of the bank to be created. It is also worth mentioning that static transaction costs should be also minimized by setting up appropriate operating protocols and boosting the use of ICT (integrating operations applications with public water records databases), geographic information systems (GIS) and remote sensing techniques. This would allow the following procedures to be streamlined: i) the presentation of offers/demand by stakeholders; ii) verification of the information they provide (e.g., water rights held, location of abstraction, consumption and effective use of the rights in recent years); iii) approval of operations and the execution of the corresponding financial transfers; and iv) compliance (e.g., ensuring that users who transfer their rights do not subsequently use them).

Well-designed water banks, as well as any other water policy, must improve transparency by making all market information available to the public (Santato *et al.*, 2016). Such information includes the parties involved, prices and trading volumes agreed, terms of the offer, etc. This information should be made public in real time through the websites of the basin authorities. In addition, the managing organizations of these banks should publish an annual report of activities which details the effective contribution they make to water governance, as in fact some of them already do.

All water banks should be self-financing as promoted by some water legislations such as the EU's Water Framework Directive (Martin-Ortega, 2012), where the principle of full cost recovery is a key issue. This means that the prices paid by buyers must cover not only the compensation required by the seller, but all operational and management costs relating to transactions, including water conveyance costs if transportation is needed. In this way, even public initiative water banks can avoid any possible hidden subsidies to water users (e.g., irrigators).

Logically, legislative reform in the regulation of water banks would be needed in order to implement the proposed improvements. In this regard, it is recommended that such reforms are carried out during a normal hydrological period as a way to plan ahead (with the necessary time, analysis and debates) for future shortages.

Lastly, it worth commenting that available (published) information about the performance of different water banks varies widely and is mainly qualitative. This has made it difficult to further analyse the existing case studies and make more consistent comparisons. We thus recommend that an international institution actually involved in water policy issues, such as the United Nations Environment Programme (UNEP) or The World Bank, launch an initiative to build and periodically update a database covering the main qualitative and quantitative features of water demand-side instruments (including water banks) implemented in each associated country. Through such an initiative, it could be feasible to ask different countries for official data on a voluntary basis. There is no doubt that it would make a positive contribution to the objectives of these international institutions, since the availability of this official information could support sounder water policy decision-making worldwide.

4.6. Chapter references

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Capítulo 5
Sharing a river: Potential performance of a water bank for reallocating irrigation water

Sharing a river: Potential performance of a water bank for reallocating irrigation water[7]

Abstract

This paper presents an ex-ante policy analysis of the implementation of a publicly run active water bank operating at the basin level designed to temporarily reallocate water resources between farmers considering different scenarios of reduced water availability (cyclical scarcity due to droughts). For this purpose, the Guadalquivir River Basin, located in southern Spain, is used as a case study. Fifteen representative farm types were considered to simulate water trading through public tender for purchasing and selling temporary water rights. The model is built at the basin level to estimate the aggregate demand and supply curves to establish expected exchange prices, volumes of water traded, enhancement in economic efficiency and improvement in rural development as measured by employment generation. The simulation results show that the proposed water bank encourages water transfers from 19% of the total water used in the case of a moderate drought to almost 40% in the case of an extreme drought, significantly reducing the economic and labor demand losses due to water shortages. The public water agency can recover all of the incurred water bank operation costs by implementing a €0.01/m³ price differential between purchase and sale prices without meaningfully affecting the performance of the water bank. Thus, we conclude that the implementation of this kind of water bank during droughts would be useful in mitigating negative effects of droughts. Thus, policymakers are encouraged to create water banks as an effective instrument to cope with droughts.

Keywords: Water Banks, Modeling, Irrigated agriculture, Guadalquivir River Basin, Spain

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5.1. Introduction

Climate change is causing a progressive reduction in water availability in many semiarid regions worldwide, as is the case in the Mediterranean region (IPCC, 2014). This fact, combined with population growth and the rising demand for food (and ultimately for irrigation water), is a primary reason for why water resources have become scarcer in these regions throughout the past few decades. In addition to the resulting increase in structural water scarcity, climate change is also producing more frequent and severe drought periods, resulting in more recurrent and intense episodes of cyclical water scarcity.

Due to the competitive advantages of irrigated versus rain-fed agriculture in these semiarid regions, the primary solution that has been advanced by public and private initiatives has been to increase water availability by building dams and other water infrastructure. This process, commonly known as supply-side water policy, was implemented during the 20th century, during which a great amount of water infrastructure was built. However, there is evidence from around the world that this kind of water policy cannot be further developed in these regions since in many river basins, new increases in water availability are technically infeasible or economically unaffordable, which is a situation called 'basin closure' (Molle *et al*., 2010). When basin development reaches the closure stage, any new water demand must be satisfied by reducing other existing water use. Under these circumstances, demand-side water policy instruments such as water trading instruments are considered to be the most suitable solutions to provide the necessary flexibility in water rights systems, allowing for a more efficient reallocation of water resources. Thus, water trading instruments are useful tools for managing both cyclical and structural scarcity.

Water trading instruments encompass a full range of institutions that facilitate voluntary exchanges of water between users (Delacámara *et al*., 2015). These markets can take different forms depending on key variables that define their operational rules (Griffin, 2016), such as the rights being traded (permanent rights, temporary rights, and options on temporary rights) or the parties allowed to trade (sellers and buyers). Regarding the latter, it is important to distinguish between 'water markets' that involve only private parties, where buyers and sellers interact directly to negotiate the terms of water rights transfers (sometimes with the participation of intermediaries or brokers), and the so-called 'water banks', which operate in a more institutionalized context where an administrative agency (public or private) acts as a necessary intermediary in the trading of rights (Rey *et al*., 2014).

Water banks are intermediaries that centralize the purchases and sales of water rights, acting between buyers and sellers (Spulber and Sabbaghi, 1994). These banks are typically managed by a public institution (e.g., water agencies). In such cases, water is transferred under the supervision of the public administration, which verifies that the water transactions fulfill all legal requirements, sometimes including constraints that are linked to environmental and social criteria (Garrido *et al*., 2012). These institutional arrangements are designed to cope with both structural scarcity (permanent exchange of water rights) and with cyclical scarcity (temporary water rights transfers).

Montilla-López *et al*. (2016) reviewed international experiences with water banks and demonstrated the advantages of this instrument over other kinds of water trading instruments (i.e., water markets). More concretely, the authors show how water banks allow for a more flexible and efficient reallocation of water resources because they facilitate contact and negotiation between buyers and sellers and they improve transparency by providing public information on prices and quantities, resulting in lower trade operation transaction costs (Garrick *et al*., 2013), thus boosting market activity and fostering a more efficient use of water resources (Grafton *et al*., 2011). Furthermore, water banks encourage government oversight of environmental and social externalities that arise from water trading. They also allow operations with environmental purposes (public offers to purchase rights without subsequent reallocation) in order to increase river flows, restore overexploited groundwater bodies, etc. (Clifford *et al*., 2004).

Numerous empirical works have focused on water markets worldwide, and many have analyzed the potential and actual performance of this instrument (Easter and Huang, 2014; Maestu, 2013). For *ex-ante* analyses of the performance of water markets, simulation models that are developed with mathematical programming are typically used (e.g., Gómez-Limón and Martínez, 2006; Garrido and Calatrava, 2009; Qureshi *et al*., 2009; Kahil *et al*., 2015), providing evidence of the potential impacts of water markets on the economy (economic efficiency), the environment (water use and other environmental issues) and society (regional development). These studies note that trade of water entitlements (permanent rights) and water allocations (temporary rights) improves the efficiency of water use at the basin level, with farmers typically playing central roles in the process.

Despite these advantages of water banks over other water trading instruments, there is little literature with a similar purpose focused on water banks. The only exceptions worth noting are the works of Qureshi *et al*. (2007), Mainuddin *et al*. (2007) and Dixon *et al*. (2012) in Australia; Medellín-Azuara *et al*. (2013) in the western United States; and Martínez-Granados and Calatrava (2014) and Pérez-Blanco and Gutiérrez-Martín (2017) in Spain. However, all of these studies simulated water banks that were designed to reduce overall water consumption in over-allocated basins for environmental reasons. Thus, empirical evidence has focused only on water banks that bought water rights in order to restore water balances (known as 'buyback'). None of these works have analyzed the implementation of water banks as instruments for reallocating water rights between productive users (e.g., between irrigators) as an alternative to other kinds of water trading instruments (i.e., water markets). This paper aims to bridge this knowledge gap by simulating the potential performance of a water bank that is designed to reallocate water resources between irrigators to check whether this is really a useful approach for coping with droughts. Thus, the objective of this work is to perform an *ex-ante* policy analysis of the implementation of water banks that trade temporary water rights (first buying these rights and then selling them to other productive users), accounting for different future reduced water availability scenarios (cyclical scarcity). For this purpose, a simulation model based on mathematical programming is built to estimate the aggregate demand and supply curves to establish the expected exchange prices, volumes of water traded, enhancement in economic efficiency and employment generation. This model was used to simulate the performance of the water bank proposed considering the irrigation sector within the Guadalquivir River Basin (GRB) in southern Spain as an illustrative case study. Although there are no previous modeling exercises that simulate water banks in this basin, there are several empirical studies that analyze the potential performance of water markets using this simulation approach (Garrido, 2000; Arriaza *et al*., 2002; Calatrava and Garrido, 2005). These previous works would provide a basis for an interesting discussion regarding the implementation of both water trading instruments.

To achieve the abovementioned objective, the remainder of the paper is organized as follows: The next section justifies the type of water bank that is proposed to improve cyclical scarcity management within the irrigation sector as the specific instrument to be simulated. The third section introduces the case of the irrigation sector in the GRB, for which an empirical implementation is developed. Section 4 details the simulation model that was developed to simulate the performance of the water bank that was proposed for enhanced drought management. The results of the simulations are summarized in Section 5. The final section concludes by providing the main insights derived from this study.

5.2. Water banks for managing drought periods within the irrigation sector

As mentioned above, the term 'water banks' covers a wide variety of institutional designs. Montilla-López *et al*. (2016) identified a number of different types of water banks, as shown in Table 5.1.

Having compared the different kinds of water banks, there is no doubt that all designs could be useful in reducing the operational transaction costs for all agents, thus boosting market activity. However, public and active water banks are assumed to improve the management of cyclical and structural water scarcity since they can exercise more effective control over market operations (reducing environmental and social negative externalities). Moreover, considering that the main purpose of the bank proposed is to reallocate water within the agricultural sector during drought periods, it is also evident that the best design for this instrument should consider the water itself (spot market) or temporary water rights (lease market) as assets to be exchanged, and all irrigators in the basin as agents who may potentially participate in market activities (purchases and sales).

In this sense, an active water bank seeks to set the conditions for the purchase and sale of rights for reallocation purposes in order to achieve a balanced market. Thus, the bank should first act as the sole water buyer of water rights (monopsony market) by organizing public water rights purchase offers and subsequently act as the unique water seller (monopoly market) of all the rights that were previously bought by organizing public sale offerings.

Table 5.1. *Water bank typology*

Variable	***Water bank typology***
Nature of the institution responsible	*Public water banks*: organized and managed by a public administration. *Private water banks*: organized and managed by means of a private initiative, generally run by non-profit organizations (NGOs).
Type of rights exchanged	*Permanent*: buy or sell water entitlements of water use rights. *Temporary* or *spot*: temporary transfer (*lease*) of water use rights or specific quantities of water (*spot*). *Option contracts*: provide buyers with the option to buy specific quantities of water in the future.
Purpose	*Reallocation of resource*: from lower-value to higher-value uses. *Environmental purposes*: purchase rights without subsequently reallocating them. *Managing risk related to water availability*.
Management strategy	*Passive*: act as an intermediary for purchases and sales (broker), either as a clearinghouse or through sealed bid double auctions. *Active (market-maker)*: adopt a proactive strategy. The purchasing system can vary depending on the nature of the public offerings. The bank may: i) establish the maximum amount of net purchases; ii) fix a maximum volume of water to be acquired; or iii) fix the market price for acquisitions.

Source: Montilla-López *et al*. (2016).

To date, there have been many experiences with a number of different water bank designs around the world. Some of these experiences are based on the same type of water bank that is proposed in this paper for ex-ante policy analysis, with most in the western states in the US. The most well-known program is likely the Drought Emergency Water Bank that was developed in California in 1991 to improve water management during a cyclical scarcity period. This water bank was designed as an active management bank, aiming to facilitate temporary transfers of water from the agricultural sector to urban use at a price set by the state government (Lund *et al*., 1992). Other similar active water banks aiming to reallocate water resources include the Colorado West Slope Bank, which has been in operation since 2009, and the Dungeness Water Exchange in Washington State, which has been active since 2013 (Montilla-López *et al*., 2016).

The Spanish Water Act was reformed in 1999, for the first time allowing the implementation of water markets and water banks for exchanging temporary water rights. Regarding the latter, the reformed act established regulations for the creation and operation of Water Exchange Centers (WEC) emulating the Californian Drought Emergency Water Bank (Embid, 2013). Per this new regulation, water banks in Spain can be established only under "exceptional situations of water scarcity" (special drought situations or severe overexploitation of aquifers) and are operated by river basin authorities (*confederaciones hidrográficas*). Once established, these public drought water banks must operate by actively buying and selling temporary water rights in order to achieve two objectives: improve efficiency in water use (reallocate water resources between productive users) and restore endangered water balances (environmental purposes or buybacks). This information confirms that the type of water bank proposed here for empirical analysis could be created immediately in Spain with the current legal framework.

Although spot water markets and water banks were legally approved two decades ago, their implementation has been rather disappointing as during extreme scarcity situations, trading activity when considering both instruments accounted for less than 5.0% of total water use, and only a quarter of these operations were accomplished through water banks (Palomo-Hierro *et al*., 2015). In fact, to date, only three WECs have been created in Spain (Guadiana, Júcar and Segura basins), all in the southeastern part of the country (where water resources are scarcer) during the drought period from 2005-2008, with the main purpose of coping with environmental problems (Montilla-López *et al*., 2016; Martínez-Granados and Calatrava, 2014; Carmona *et al*., 2011). In the GRB, another closed basin in southern Spain, the national government also approved the creation of a WEC in 2005, but in the end, this water bank was not actually implemented.

Because of the advantages of water banks in managing water shortages from droughts and improving efficiency in water use while minimizing environmental and social negative externalities from trade activities, Spanish river basin authorities should reconsider the role of this instrument in their policy mix. Thus, this paper aims to provide information on the potential performance of a water bank for reallocation purposes.

5.3. Case study

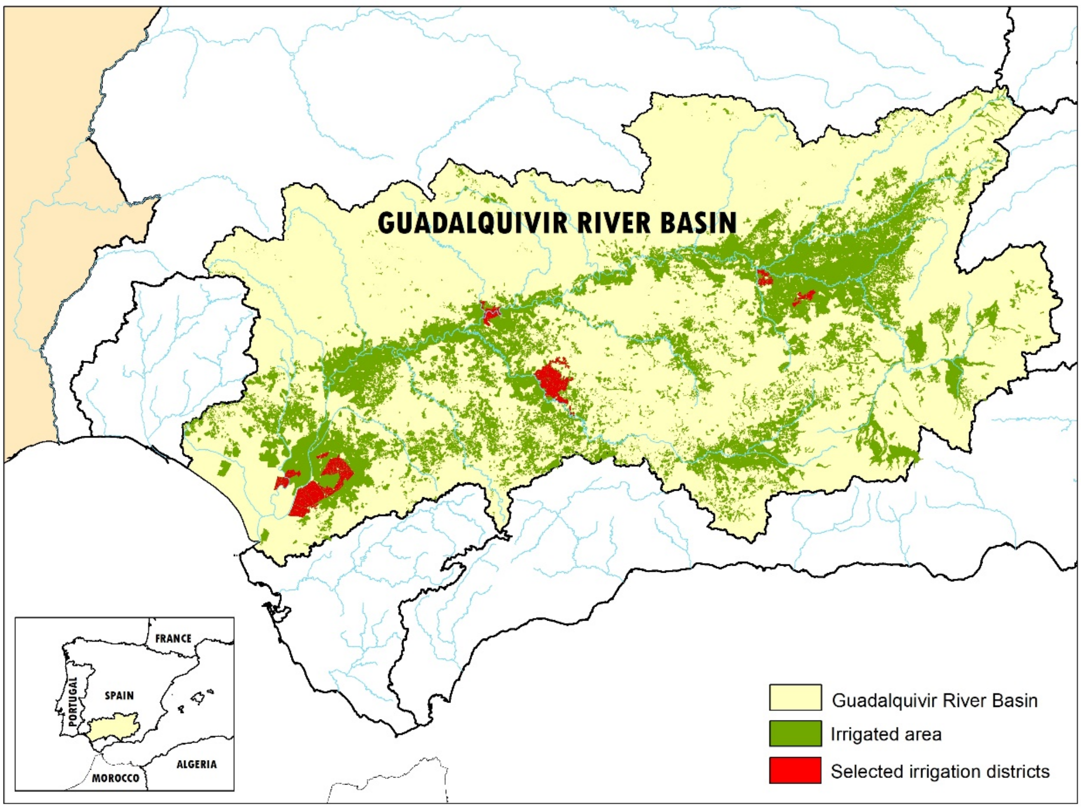
5.3.1. The Guadalquivir River Basin

The case study focuses on the GRB, a semiarid region located in southern Spain (see Figure 5.1) that covers a total surface area of 57,184 km^2 and is home to a population of 4.4 million inhabitants. The GRB has a typical Mediterranean climate with hot and dry summers and mild winters and frequent episodes of hydrological drought.

As in many other regions worldwide, the GRB is currently closed to new users due to a significant increase in water demand over the past few decades, mainly due to the growth of irrigated areas, which currently cover approximately 28% of the agricultural area in the basin (Expósito and Berbel, 2017). Thus, demand-side management has become the only tool available for managing new water demands.

The average water use in the GRB amounts to 3,815 Mm^3 per year, of which approximately 3,357 Mm^3 is used for agricultural irrigation (88% of the total water demand) and 379 Mm^3 is used by households and for other urban demands (10%) (CHG, 2015). Spanish water law deems that urban users have senior water rights; this is, their water demands are served first in the event of water scarcity. This priority system leads urban users to normally be unwilling to participate in either water markets or banks since their water demands are always covered. Thus, considering that irrigated agriculture accounts in this basin almost all water users potentially willing to participate in water trading, only irrigators are considered operating agents when simulating the water bank proposed here. Moreover, it is worth pointing out that water trading among irrigators is technically feasible because most irrigation districts in the basin are located in the same water management system, allowing water transfers to use existing infrastructure.

Figure 5.1. *Location of selected irrigation districts in the Guadalquivir River Basin*



Source: own elaboration

The potential performance of water markets in the GRB is high because of the significant differences in the marginal values of water in several irrigation districts and farm types within the basin, as shown by Garrido (2000), Arriaza *et al*. (2002) and Calatrava and Garrido (2005). However, the actual development of water trading in this basin has been very limited. In fact, a very low quantity of water has been traded through spot water markets, and water banks have not yet been implemented. For instance, in the busiest market year (2007, during which there was a severe drought), 33.2 Mm^3 were transferred that corresponded only to 0.88% of the total water use. Moreover, most of the water transfers had a different basin as a destination, 20 Mm^3 was allocated to the irrigation districts in the Andalusian Mediterranean Basins

(Garrido *et al*., 2012; Palomo-Hierro *et al*., 2015). The main reasons behind the disappointing performance of these reallocation mechanisms in the GRB are twofold. On one hand, the activity of spot water markets has been very constrained due to legal, administrative, cultural, psychological, and technical barriers (Palomo-Hierro *et al*., 2015; Giannoccaro *et al*., 2016). These barriers have ultimately led to high transaction costs that in turn have limited the number of mutually beneficial transactions. On the other hand, water banks have not yet been used in this basin to reallocate water resources due to the lack of political will to implement such a plan (Montilla-López *et al*., 2016).

In this sense, the motivation of this paper is to propose a water bank capable of minimizing abovementioned barriers in order to improve water management by boosting water trade. In fact, it can be assumed that most legal and administrative barriers would be removed because is precisely the public agency in charge of bank management that would be promoting transactions, so they will be expected to be minimized. Moreover, cultural and psychological barriers would be also minimized, because: a) selling (or buying) water to (or from) the public agency would avoid the negative idea of gaining a profit from common resources granted by that same public agency, and b) a publicly run water bank provides more legal security to willing buyers and sellers that ultimately could reduce psychological barriers.

5.3.2. Data acquisition

To properly model the potential performance of a water bank as proposed by Spanish law, the heterogeneity of the irrigation districts located in the GRB should be considered. Gómez-Limón *et al*. (2013) classified irrigation districts in this basin into the following five categories: C1: "Traditional annual crops" (4.6% of total irrigated area in the GRB), C2: "Modernized irrigated areas" (36.9%), C3: "Modern olive groves" (48.3%), C4: "Traditional vegetables" (6.1%), and C5: "Traditional rice" (4.1%). Considering this classification, we selected seven representative irrigation districts (see location in Figure 1) that account for a total area of 49,562 hectares (5.8% of total irrigated area in the basin).

C3 is the largest type among the irrigated areas, though this category is very homogeneous since all of these regions are devoted to only olive growing, which justifies modeling this category by considering only two irrigation districts: *Pajarejos* and *Santiago Apóstol*. C2 is the most heterogeneous category in terms of farming systems, and as such, three districts within this type have been selected for inclusion in the basin model: *Genil-Cabra*, *Sector BXII del Bajo Guadalquivir* and *Bembézar Margen Izquierda*. C1 and C5 are quite small and homogeneous categories that can be represented by only one irrigation district: *Marismas del Guadalquivir* and *La Ermita*, respectively[8]. The main characteristics of the selected irrigation districts are shown in Table 5.2.

Table 5.2. *Selected irrigation districts in the Guadalquivir River Basin*

Irrigation district	*Irrigated area (ha)*	*Farms (No.)*	*Main crops (%)*	*Water use (m^3/ha·year)*	*Irrigation technology (%)*
Las Marismas del Guadalquivir (type C1)	11,980	850	Corn (10%), Alfalfa (16%), Cotton (63%)	6,100	Farrow (100%)
Genil-Cabra (type C2)	15,500	1,563	Wheat (20%), Sunflower (22%), Olive groves (35%)	2,600	Sprinkler (60%), Drip (40%)
Sector BXII del Bajo Guadalquivir (type C2)	14,654	499	Tomato (19%), Sugar beet (21%), Cotton (44%)	6,000	Farrow (4%), Sprinkler (75%), Drip (21%)
Margen Izquierda del Bembézar (type C2)	4,009	163	Orange groves (48%) and Corn (28%)	6,000	Sprinkler (12%), Drip (88%)
Pajarejos (type C3)	2,262	490	Olive groves (100%)	1,500	Drip (100%)
Santiago Apóstol (type C3)	2,650	790	Olive groves (100%)	1,500	Drip (100%)
La Ermita (type C5)	4,306	243	Rice (100%)	11,500	Farrow (100%)

After selecting the abovementioned irrigation districts, the primary information to input into the simulation model was obtained from direct surveys that were conducted in spring 2014, involving managers of each district and 355 farmers, and

[8] C4 is not specifically represented in the basin model because of its low relevance in terms of irrigated area and the volume of water used and because its geographic location makes it difficult to exchange water resources due to technical barriers to trade.

were sampled using random routes along the irrigation districts under consideration. The questionnaires provided information about farm size, crop mix, irrigation technology and water use, among other descriptive variables about farms in the different irrigation districts. Secondary information was obtained from official agricultural statistics to generate historical time series of income, revenues and profitability indicators for the different crops in each district for the period 2007-2013.

Real-life observations show that there is heterogeneity between farms regarding crop mixes, agricultural practices and water use within the same irrigation district, mainly due to differences in farmers' objectives (Berkhout *et al*., 2011). Thus, considering irrigation districts as units to be modeled would lead to serious aggregation bias. As such, for modeling purposes, sufficiently homogeneous farm types are usually considered decision units to minimize this aggregation bias. To define these homogenous groups of farms within each irrigation district, statistical clustering techniques were used. From the set of such clustering techniques, we selected Euclidean distance as the measure of distance between farms exemplifying features (crop mix), and we selected Ward's method as the criterion for aggregation (Hair *et al*., 2010). Following this procedure, different groups of farms (clusters) were obtained in each irrigation district with the exception of homogeneous districts with only a single crop (those representing olive groves or rice systems), where all farms were included in the same group (Berbel and Rodríguez-Ocaña, 1998; Gómez-Limón and Riesgo, 2004). The resulting clusters of farms were characterized by calculating the average values of the different variables that were collected, including crop mix and other variables related to farm and farmer features (farm size and irrigation technology, farmer's age, gender, educational level and agricultural training). The average values for these variables were used to define the corresponding farm types, as shown in Table 5.3. These farm types were used as decision units in the model in order to simulate water trading through the water bank.

Table 5.3. *Irrigated farm types in the Guadalquivir River Basin*

Irrigation district	*Farm types*	*"Label": main crops (%)*	*Age (years)*	*Farm size (ha)*	*Income from agriculture / Total income*
Las Marismas del Guadalquivir	MG1	"*Cotton growers*": Cotton (97%), Sunflower (3%)	53.2*	13.6	82.0%
	MG2	"*Large farmers with a commercial profile*": Cotton (61%), Corn (19%), Sugar beet (7%), Sunflower (7%)	52.0*	35.6	80.0%
	MG3	"*Conservative small farmers*": Cotton (43%), Alfalfa (57%)	46.8*	18.0	82.0%
Genil-Cabra	GC1	"*Large diversified farmers*": Olive groves (31%), Sunflower (27%), Wheat (20%), Corn (8%), Cotton (7%)	56.0	35.0***	63.0%**
	GC2	"*Traditional mixed farmers*": Wheat (25%), Olive groves (24%), Sunflower (20%), Cotton (14%), Corn (10%)	53.5	25.8***	56.9%**
	GC3	"*Part-time irrigation olives growers*": Olive groves (100%)	53.7	6.6***	24.5%**
Sector BXII del Bajo Guadalquivir	BG1	"*Large professional farmers*": Tomato (30%), Cotton (30%), Sugar beet (24%), Vegetables (7%)	52.7	35.8**	90.0%
	BG2	"*Risk diversifying farmers*": Cotton (57%), Tomato (13%), Corn (9%), Wheat (7%), Sugar beet (6%)	54.6	23.9**	95.0%
	BG3	"*Extensive conservative farmers*": Cotton (57%), Sugar beet (39%), Wheat (4%)	57.8	15.0**	95.8%
Margen Izquierda del Bembézar	MIB1	"*Large diversified professional farmers*": Corn (32%), Orange groves (25%), Olive groves (12%), Sunflower (8%), Wheat (7%), Cotton (6%), Vegetables (5%)	54.7	79.65**	73.6%
	MIB2	"*Citriculture traders*": Orange groves (100%)	47.9	47.6**	65.5%
	MIB3	"*Small part-time corn growers*": Corn (100%)	47.4	13.1**	42.9%
Pajarejos	PAJ	"*Olive growers*": Olive groves (100%)	56.2	17.4	64.3%
Santiago Apóstol	SA	"*Olive growers*": Olive groves (100%)	57.6	6.4	37.3%
La Ermita	LE	"*Rice growers*": Rice (100%)	54.4	39.3	64.9%

*, **, and *** denote significance at the 5, 1, and 0.1% levels, respectively from the ANOVA test comparing cluster average values within the same irrigation district.

5.4. Modeling approach

5.4.1. Farmers' decision making: optimizing Cobb-Douglas MAUFs

Classical economic theory relies on the assumption that farmers' behavior can be modelled by maximizing profits or any utility function with profits as a single-attribute. However, farmers' decision-making processes are driven by various, usually conflicting criteria, in addition to the expected profit. In this way, it can be assumed that producers' decision making is guided by the maximization of a multi-attribute utility function (MAUF), where all relevant attributes that are considered are condensed, which is the main idea that underlies Multi-Attribute Utility Theory (MAUT), an approach that was largely developed after the publication of the seminal work by Keeney and Raiffa (1976) to overcome the limitations of single-attribute (profit related) utility functions. This alternative approach has also been widely implemented for simulating farmers' behavior, as shown in Sumpsi *et al*. (1997), Amador *et al*. (1998), Gómez-Limón and Berbel (2000) and Gómez-Limón *et al*. (2004), among others.

Given that linear specifications for MAUFs are easier to elicit and interpret, most empirical implementations of the MAUT approach for simulating farmers' decision making have relied on the elicitation of additive MAUFs, usually estimated with a non-interactive procedure based on weighted goal programming, as suggested by Sumpsi *et al*. (1997). However, it is worth pointing out that considering additive MAUFs implies linear indifference curves (also called iso-utility curves or iso-preference curves), a condition that is somewhat restrictive because it involves the oversimplified behavior of real decision makers. This implication also makes additive MAUFs inaccurate when simulating actual decision making (Hardaker *et al*., 2007).

These limitations have encouraged authors to use more general and flexible multiplicative forms for MAUFs since these types of utility functions allow more real indifference curves. In fact, as has been shown by André and Riesgo (2007), the application of multiplicative utility functions could be more successful in reproducing farmers' behavior than additive ones. Following this line of research, we aim to simulate farmers' decision-making process using the Cobb-Douglas utility function as has been proposed by Gutiérrez-Martín and Gómez-Gómez (2011) and Gómez-Limón *et al*. (2016). This choice is justified because this function is coherent with neoclassical economic theory since it guarantees that there is a global optimum when the efficient frontier is convex and because this formulation is consistent with the postulate of decreasing marginal utility for every attribute.

The proposed formulation of the Cobb-Douglas MAUF is as follows:

$$U(\boldsymbol{X}) = \prod_{a} u_a\big(f_a(\boldsymbol{X})\big)^{\alpha_a} \tag{1}$$

where $\boldsymbol{X}(n \times 1)$ is the vector of decision variables, X_c is the area devoted to each productive activity (crop), $u_a\big(f_a(\boldsymbol{X})\big)$ is a single-attribute or partial utility function related to attribute a, and α_a denotes the coefficient of each attribute that expresses its relative importance. These coefficients are assumed to be lower than one in order to ensure decreasing marginal utility for every attribute. Furthermore, these coefficients are assumed to sum to 1 for normalization purposes.

Based on the abovementioned consideration, farmers' productive behavior can be simulated using a mathematical programming model in which the utility function (1) is the objective function to be maximized:

$$Max\ U(\boldsymbol{X}) = \prod_{a=1}^{m} \big[u_a\big(f_a(\boldsymbol{X})\big)\big]^{\alpha_a} \tag{2.1}$$

$$\text{s.t.} \quad \sum_{a=1}^{m} \alpha_a = 1 \tag{2.2}$$

$$\boldsymbol{AX} \leq \boldsymbol{B} \tag{2.3}$$

$$\boldsymbol{X} \geq 0 \tag{2.4}$$

where $U(\boldsymbol{X})$ represents a farmer's utility in a multi-attribute setting, which depends on a set of m single or partial utility functions ($u_a\big(f_a(\boldsymbol{X})\big)$) considering all relevant attributes ($f_a(\boldsymbol{X})$) for a producer's decision making. As mentioned above, the MAUF proposed as the objective function is a homothetic Cobb-Douglas function; thus, the objective function needs to be constrained by expression (2.2). Finally, the model constraints are built based on matrix $\boldsymbol{A}$ $(p \times n)$ of technical coefficients in allocable resource constraints and vector $\boldsymbol{B}$ $(p \times 1)$ of available resource levels.

5.4.2. Eliciting farmers' MAUF

To simulate the farmer's decision-making process under a Cobb-Douglas MAUF, it is necessary to elicit the values of the calibrating parameters α_a. For this purpose, the values that most closely approximate the observed behavior are found, following a similar procedure to that developed by Sumpsi *et al*. (1997), as explained below.

For operational purposes, we consider that (i) all relevant attributes are related to the objectives that are to be maximized (i.e., more-is-better attributes), and (ii) each single-attribute or partial utility function $(u_a(f_a(\boldsymbol{X})))$ is equal to the corresponding attribute and is properly normalized to be between 0 and 1 $(nf_a(\boldsymbol{X}))$. Thus, the crop mix selection $(\boldsymbol{X})$ can be seen as a multi-objective programming (MOP) decision-making problem (Gómez-Limón *et al*., 2016). MOP problems seek to obtain the Pareto-efficient subset from the feasible solutions (election-possibility set, denoted by F), assuming that whatever preferences decision makers may have, their choices belong to the efficient frontier. A first approximation to this efficient frontier can be assessed through the pay-off matrix. This matrix is obtained by maximizing each of the objectives (i.e., partial utility functions) subject to the constraints set (expressions 2.2 to 2.4).

In mathematical terms, one of the advantages of the Cobb-Douglas function is its potential to be transformed into the additive function, which is as follows:

$$log[U(\boldsymbol{X})] = V(\boldsymbol{X}) = \sum_{a=1}^{m} \alpha_a \cdot log[nf_a(\boldsymbol{X})] \tag{3}$$

This transformation allows for following a similar procedure to that developed by Sumpsi *et al*. (1997) to estimate more appropriate alpha parameters by solving the following m+1 system of equations:

$$\begin{bmatrix} log(nf_{11}) = \log(nf_1^*) & log(nf_{12}) & \cdots & log(nf_{1m}) \\ log(nf_{21}) & log(nf_{22}) = log(nf_2^*) & \cdots & log(nf_{2m}) \\ \vdots & \vdots & \ddots & \vdots \\ log(nf_{m1}) & log(nf_{m2}) & \cdots & log(nf_{mm}) = log(nf_m^*) \end{bmatrix} \cdot \begin{bmatrix} \alpha_1 \\ \alpha_2 \\ \vdots \\ \alpha_m \end{bmatrix} = \begin{bmatrix} log(nf_1^{obs}) \\ log(nf_2^{obs}) \\ \vdots \\ log(nf_m^{obs}) \end{bmatrix} \tag{4.1}$$

$$\sum_{a=1}^{m} \alpha_a = 1 \tag{4.2}$$

where nf_a^* is the normalized ideal value for attribute a, $nf_{aa'}$ are the normalized values of the elements in the pay-off matrix of attribute a when attribute a' is optimized, and nf_a^{obs} are the normalized observed values of each attribute.

Usually, there is not an exact solution to the above system, and it is therefore necessary to solve the problem by minimizing the sum of the deviational variables that find the closest set of parameters α_a:

$$\min \sum_{a=1}^{m} (n_a + p_a) \tag{5.1}$$

s.t.

$$\alpha_1 \cdot log(nf_{11}) + \alpha_2 \cdot log(nf_{12}) + \cdots + \alpha_m \cdot log(nf_{1m}) + n_1 - p_1 = log(nf_1^{obs}) \tag{5.2}$$

$$\alpha_1 \cdot log(nf_{21}) + \alpha_2 \cdot log(nf_{22}) + \cdots + \alpha_m \cdot log(nf_{2m}) + n_2 - p_2 = log(nf_2^{obs}) \tag{5.3}$$

$$\cdots \qquad \cdots$$

$$\alpha_1 \cdot log(nf_{m1}) + \alpha_2 \cdot log(nf_{m2}) + \cdots + \alpha_m \cdot log(nf_{mm}) + n_m - p_m = log(nf_m^{obs}) \tag{5.m+1}$$

$$\sum_{a=1}^{m} \alpha_a = 1 \tag{5.m+2}$$

where n_a and p_a are the absolute negative and positive deviations, respectively.

5.4.3. MAUF elicitation for the case study

To model farmers' decision making, we selected the three most relevant attributes for irrigators in the case study considered: (i) the expected total gross margin as a proxy of profit in the short run ($f_1(\boldsymbol{X}) = GM(\boldsymbol{X})$), to be maximized; (ii) the production risk measured as the variance of the gross margin ($f_2(\boldsymbol{X}) = VAR(\boldsymbol{X})$), to be minimized; and (iii) the total labor ($f_3(\boldsymbol{X}) = TL(\boldsymbol{X})$) as a proxy of managerial complexity, also to be minimized. The concrete formulations of these attributes are explained in the next section. Each of these attributes are defined as a mathematical function of decision variables, the area covered by alternative productive activity ($\boldsymbol{X}$). In our case study, these activities are denoted by $X_{i,j}$, where i means the crop and j indicates the irrigation technique used.

As explained above, for operational purposes, partial utilities functions must be normalized in order to transform them into more-is-better and dimensionless functions with values varying within the interval [0,1] ($nf_a(\boldsymbol{X})$). Aiming to fulfil these requirements, we propose transforming the original attribute functions into rates of success with respect to the ideal value of each attribute (GM^*, VAR^* and TL^*, respectively). Thus, the normalized attributes can be represented as follows:

$$nf_{GM}(\boldsymbol{X}) = \frac{GM(\boldsymbol{X})}{GM^*}; \quad nf_{VAR}(\boldsymbol{X}) = \frac{VAR^*}{VAR(\boldsymbol{X})}; \quad nf_{TL}(\boldsymbol{X}) = \frac{TL^*}{TL(\boldsymbol{X})} \tag{6}$$

Note that more-is-better attributes, such as $GM(\boldsymbol{X})$, are normalized differently than less-is-better attributes, such as $VAR(\boldsymbol{X})$ or $TL(\boldsymbol{X})$; for the former, the ideal values (the largest possible values) are in the denominator, and for the latter, the ideal values (the smallest possible values) are in the numerator. Thus, it can be checked that by proceeding in this way, all normalized attributes are related with more-is-better objectives and that their values range between 0 and 1.

Thus, the shape of the MAUF to be used for modeling purposes is:

$$U(\boldsymbol{X}) = nf_{GM}(\boldsymbol{X})^{\alpha_{GM}} \cdot nf_{VAR}(\boldsymbol{X})^{\alpha_{VAR}} \cdot nf_{TL}(\boldsymbol{X})^{\alpha_{TL}} \tag{7}$$

where the alpha parameters are estimated by running model (5).

5.4.4. Modeling farmers' decision making considering water bank operations

To simulate water bank operations, it is necessary to know how farmers would react if they could sell or purchase water through this kind of water market. Thus, in addition to variables $\boldsymbol{X}$, it must be considered that farmers can also make decisions regarding the quantity of water sold to the bank (or water bank purchases, denoted by WBP) or the quantity of water purchased from the bank (or water banks sales, denoted by WBS). Considering these two new decision variables, farmers' decision-making process in this setting can be simulated by the following model:

$$max\ U(\boldsymbol{X}) = nf_{GM}(\boldsymbol{X})^{\alpha_{GM}} \cdot nf_{VAR}(\boldsymbol{X})^{\alpha_{VAR}} \cdot nf_{TL}(\boldsymbol{X})^{\alpha_{TL}} \tag{8.1}$$

$$GM(\boldsymbol{X}) = \sum_i \sum_j \{(p_i \cdot y_{i,j} + s_i - vc_{i,j}) \cdot X_{i,j}\} + wp_p \cdot WBP - wp_s \cdot WBS \tag{8.2}$$

$$VAR(\boldsymbol{X}) = \boldsymbol{X}^t \cdot [\boldsymbol{cov}] \cdot \boldsymbol{X} \tag{8.3}$$

$$TL(\boldsymbol{X}) = \sum_i \sum_j tl_{i,j} \cdot X_{i,j} \tag{8.4}$$

s.t.

$$\sum_i \sum_j X_{i,j} \leq fa \tag{8.5}$$

$$\sum_i \sum_j X_{i,j} \cdot w_{i,j} \leq \delta \cdot wa \cdot fa + WBS - WBP \tag{8.6}$$

$$\boldsymbol{AX} \leq \boldsymbol{B} \tag{8.7}$$

$$X_{i,j}, WBP, WBS \geq 0; \quad \forall\ i,j \tag{8.8}$$

Equations (8.2) to (8.4) are mathematical representations of partial utility functions. Thus, the farm gross margin is calculated as the sum of total income, including both product sales (crop price –p_i– multiplied by yield per crop and irrigation technique –$y_{i,j}$) and coupled subsidies (s_i) minus the variable costs from crops ($vc_{i,j}$). Furthermore, considering the operations in water markets, the gross margin also includes the income from the water sold to the bank (water price –wp_p– multiplied by the quantity of water purchased by the bank –WBP) and the cost derived from the water bought from the bank (water price –wp_s– by the quantity of water sold by the bank –WBS). Risk is calculated in equation (8.3) as the variance of the gross margin, where $[\boldsymbol{cov}]$ is the variance-covariance matrix of the gross margins of crops per hectare during the period 2007-2013. The last attribute is total labor, which is calculated in equation (8.4) as the sum of labor requirements per crop and the irrigation technique ($tl_{i,}$) for the entire farm.

Constraints (8.5) and (8.6) are related to land and water availability, respectively. The first constraint limits the total area covered by the different alternatives to the farm size (fa). The water constraint establishes that irrigation water requirements cannot exceed water availability, the former being the sum of water requirements per alternative, and the latter is the water allotment provided by the water agency considering farm size (fa), water rights granted per hectare (wa) and resource availability at the basin level (water availability coefficient, δ) plus/minus the quantity of water purchased/sold from/to the water bank (WBS–WBP).

Equation (8.7) denotes the other constraints that define the feasible solution set. For this purpose, the following restrictions were considered: (i) *agronomic practices*, allowing only those rotational practices actually followed by farmers; (ii) *permanent crops* (fruit and olive groves), fixing the area covered by each of these crops in the short run; (iii) *irrigation techniques*, constraining the area irrigated by each irrigation system (surface, sprinkler and drip) as irrigation equipment is also fixed in the short term; (iv) *sugar beet and cotton quotas*, limiting the maximum area devoted to each of these two crops because of agricultural policy production quotas; and (v) *market constraints* for perishable agricultural products such as tomatoes, garlic, onions, and carrots because of limited marketing channels. Finally, we also establish that decision variables ($X_{i,j}$, WBP and WBS) must be non-negative, as denoted by equation (8.8).

5.4.5. Simulation model for a water bank at the basin level

The potential performance of the water bank that is proposed for trading temporary water rights was simulated considering that first, the bank implements a public offer to purchase these rights, and afterwards, it sells all the rights that were previously bought through a public sale offering, with both tenders organized considering centralized trading rules (i.e., a monopsony market for right purchases and monopoly market for sales). For this purpose, a two-step procedure is implemented.

The first step is the estimation of the aggregate demand and supply curves at the basin level. These curves are estimated by aggregating the results of the models that were developed for the 15 representative farm types (see Table 5.3), accounting for the relative weight of each type over the total irrigated area at the basin level. The aggregation procedure followed for this purpose is double-sided. Firstly, farm type models were run in order to simulate the purchases of temporary water rights by the bank from willing sellers. For this purpose, the value of wp_p has been parametrized from €0.00/m^3 to €1.00/m^3. Thus, by running these models, the aggregate supply of temporary rights is estimated; that is, the total quantity of water that the bank could buy for every purchase price. Secondly, the same farm type models were run in order to parametrize the value of wp_s from €0.00/m^3 to €1.00/m^3, which allowed us to build the aggregate demand of temporary rights, that is, the total amount of water that the bank could sell to farmers that are willing to buy additional water at each sale price.

It is worth pointing out that this first step was implemented for three scenarios of water availability to simulate droughts with different values for the coefficient of water availability δ. For our case study, the values of 0.75 ('moderate' drought; only 75% of water rights are available), 0.50 ('severe' drought; only 50% of water rights available) and 0.25 ('extreme' drought; only 25% of water rights are available) were considered[9]. Figure 5.2 shows the resulting aggregate demand and supply curves in each of these scenarios.

Once aggregate demand and supply curves are estimated, the second step in the procedure is calculating the amount of water that the bank should trade. Assuming that the proposed public water bank is a non-profit and non-subsidized institution that is willing to optimize water use efficiency, its objective should be to maximize the quantity of water that is reallocated among irrigators, considering only the

[9] This simplification can be considered plausible enough for modeling purposes since most of the irrigated area in the basin are located within the same water management system, where all available water resources are shared (i.e., they have the same annual water allotment). In any case, the same modeling approach could be used considering different δ parameters for each irrigation districts if required for simulating heterogenous water availability.

constraint that $wp_p^* \leq wp_s^*$. In the absence of operational costs, this maximum is reached in every scenario of water availability when the aggregate demand and supply curves intersect. In this point, the optimum bank outcome fulfils two conditions: a) the quantity of water purchased is equal to the quantity of water sold ($q_p^* = q_s^*$), and b) purchase price is equal to the sale price ($wp_p^* = wp_s^*$). Thus, to achieve this outcome, the bank should implement a public offer to purchase temporary water rights at price wp_p^* in a monopsonistic market, buying the amount of water q_p^*. Afterward, it should organize a public offer to sell at the same price in a monopolistic market, then selling the same amount of water.

If non-null operational costs are considered, the purchase price should be lower than the sale price ($wp_p^* < wp_s^*$) in order to allow the public agency that is in charge of the bank operations to cover its costs with its own income (assumption of non-subsidized institution). In this situation, the maximization of water traded (i.e., maximum efficiency in water use) is achieved by fixing wp_p and wp_s as close to each other as possible, with the difference between both prices serving only to cover the costs of water bank operations borne by the public agency. Thus, to reach the optimum bank outcome with operational cost, we graphically looked for the gap between aggregate demand and supply whose value is equal to the operational costs considered, equaling the amount of water purchased and sold ($q_s^* = q_p^*$). The prices for aggregated supply (wp_p^*) and demand (wp_s^*) for this situation would be the prices to be used by the bank for the public offer to purchase temporary water rights in a monopsonistic market and for the public offer to sell them in a monopolistic market, respectively.

An accurate estimate of water bank operation costs for this case study is beyond the scope of this work. In any case, for illustrative purposes only, staff from the Júcar and Guadiana water agencies who have already created and managed water banks and other experts were consulted, and they roughly estimate that these costs for the Guadalquivir basin would fall within a range from 1 to 2 million euros annually. Because it is assumed that the simulated water bank is not a subsidized institution, a different water price for water purchases and sales will be applied in order to generate a positive net public revenue to cover all operational costs borne by the public agency. For this purpose, the difference $wp_p^* - wp_s^*$ has been parameterized from €0.01/m^3 to €0.03/m^3, allowing us to analyze the impact of this differential price strategy on the quantity of water traded, the economic and social effects and the public income needed to cover these costs.

It is worth clarifying that the optimal bank outcome that is reached as explained above is equivalent to the market equilibrium provided by a hypothetical perfect competitive spot water market or a passive water bank that operates through a clearing house (this kind of water bank does not use its own budget to buy and sell, but acts only as an intermediary). Thus, it is relevant to remark that the optimal reallocation of water resources from an economic point of view (maximum efficiency of water use) can be reached by several water trading instruments, including active water banks.

The bank outcome that was calculated as explained above can provide useful information about the potential economic efficiency that is reachable with this trading instrument. However, to assess this potential performance it is also relevant to consider the social impact of the instrument, which can be estimated by measuring the changes in employment generation. In our modeling approach, these changes are calculated by comparing simulated agricultural labor demand for the different drought scenarios with and without water bank reallocations. For both scenarios, labor demand is estimated based on the crop mixes that are implemented in each case. As temporary water rights are transferred from extensive and low value-added activities to more intensive and profitable crops, it is expected that water reallocation leads to an aggregated net increase in employment generation (with the labor increase in water-buying farms being higher than labor losses in water-selling farms).

5.5. Results

5.5.1. Elicited utility functions

Running the calibration procedures as explained in the model (5), the sets of calibration parameters α_a have been elicited in order to obtain the MAUFs that are to be optimized by each farm type considered. In this way, Table 5.4 shows the results obtained.

Table 5.4 also shows two indexes of calibration goodness, measuring the differences between the observed values and the simulated values when the utility function is maximized. The first index is the mean squared error (MSE), which measures these differences in the space of attributes:

$$MSE = \sqrt{\frac{\sum_{a=1}^{m}\left(\frac{f_a^{obs}(\boldsymbol{X}) - f_a(\boldsymbol{X})}{f_a^{obs}(\boldsymbol{X})}\right)^2}{m}} \tag{9}$$

Table 5.4. *Estimated values of MAUF parameters and model validation indexes*

Irrigation district	*Farm type*	α_{GM}	α_{VAR}	α_{TL}	*MSE (%)*	*FK index*
Las Marismas del Guadalquivir	MG1	1.00			0.5%	99.8%
	MG2	0.96		0.04	36.8%	67.6%
	MG3	1.00			7.5%	97.8%
Genil-Cabra	GC1	0.95		0.05	8.2%	73.3%
	GC2	0.97		0.03	5.9%	79.7%
	GC3	1.00			0.0%	100.0%
Sector BXII del Bajo Guadalquivir	BG1	0.98	0.01	0.01	9.9%	96.8%
	BG2	0.98	0.01	0.01	10.8%	96.2%
	BG3	1.00			0.0%	100.0%
Margen Izquierda del Bembézar	MIB1	0.89	0.11		7.1%	88.9%
	MIB2	1.00			0.0%	100.0%
	MIB3	1.00			0.0%	100.0%
Pajarejos	PAJ	1.00			0.0%	100.0%
Santiago Apóstol	SA	1.00			0.0%	100.0%
La Ermita	LE	1.00			0.0%	100.0%

The second is the Finger-Kreinin (FK) similarity index, which measures the differences in the space of decision variables (i.e., crop areas):

$$FK\ similarity\ index = \sum_i \sum_j min\left(\frac{X_{i,j}}{fa}; \frac{X_{i,j}^{obs}}{fa}\right) \tag{10}$$

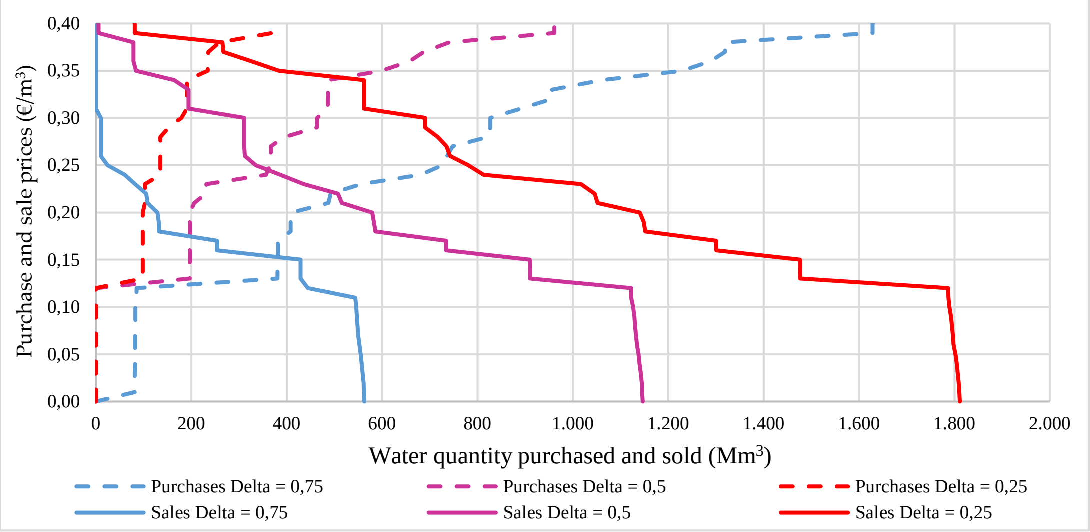
The results provided by both indexes show that calibration procedure proposed offers a reasonably good approximation of farmers' decision making.

5.5.2. Water bank performance without operational costs

This second sub-section analyzes the outcomes that were obtained from the simulations of different scenarios of water availability (the coefficient of water availability δ equals 0.75, 0.50 or 0.25) in the ideal case where operational costs of zero are considered (i.e., market balance achieved for $wp_p = wp_s$). These scenario simulations are interesting for two reasons. First, these results show the maximum potential of water banks to increase economic efficiency. Second, these simulations also display the performance of a subsidized water bank when all transactions costs are covered by the public budget.

Figure 5.2 shows the resulting aggregated curves for purchases and sales in each of three water availability scenarios. As we can see, when water availability is reduced to 75% of water rights (moderate drought; δ=0.75), the equilibrium price managed by the water bank reaches €0.15/m³, involving 381 Mm³ of water traded. For the intermediate scenario that simulates a severe drought (δ=0.50), the purchase and sale prices reach €0.24/m³, and 363 Mm³ of water is exchanged. Finally, when the drought is extreme (δ=0.25), the clearing price is €0.38/m³, and 259 Mm³ is exchanged.

Figure 5.2. *Market balance*



The water prices and quantities of water traded that were obtained for each scenario are also shown in Table 5.5. This quantitative assessment clearly shows that despite the decrease in water quantity traded in absolute terms (Mm^3) as water availability is reduced, the percentage of water exchanged over the total water used at the basin level increases as the drought becomes more acute. In fact, this percentage reaches 38.9% for the extreme drought scenario, while for the 'moderate' drought scenario, the water bank only mobilizes 19.1% of the total usable water.

These results can be compared with previous work that focused on simulating the spot water market in the same river basin without operational costs. In this sense, it is worth quoting the work of Arriaza *et al*. (2002), where potential of water markets were analyzed in a single irrigation district (*El Bajo Guadalquivir*), showing that a reduction of water availability in a 50% (annual allotment of 2,670 m^3/ha) would lead to equilibrium price of €0.15/m^3, exchanging only 4% of total water used. Similarly, Calatrava and Garrido (2005) analyzed the impact of a spot water market in another irrigation district within the GRB (*Guadalmellato*), and their estimates show that a water allotment of only 50% compared with a normal hydric year (annual allotment of 2,800 m^3/ha) would require that 21% of total water used was reallocated among irrigators at a price of €0.12/m^3. It is worth noting that both works simulated local water markets (irrigation district level). This scope is much lower than that considered in this paper (basin level), the latter of which allows much more users (higher heterogeneity in water utility) to participate in the exchanges, which could explain why the estimates reported here suggest a more active reallocation in terms of the share of water resources exchanged.

Considering the five categories of irrigation districts that are explained in Section 3, the results show that in general, the farm types located in modernized irrigation areas (type C2) are willing to purchase water when a moderate drought occurs. Some of these farms, the more profitable ones (those oriented to fruit and vegetable crops), also continue to acquire water from the water bank in severe drought situations. However, only farm types with perennial crops (fruit, citrus and olive groves) are willing to purchase water when an extreme drought is in place[10]. The farm types located in the traditional rice irrigated areas (type C5) are the main sellers in all drought situations. For severe and extreme droughts, less profitable farm types located in traditional annual crops irrigated areas (type C1) also behave as sellers to the water bank.

Table 5.5. *Water bank performance for null operational costs*

	Coefficient of water availability		
	75%	*50%*	*25%*
Water quantity used (Mm^3)	1,995	1,330	665
Water price (€/m^3) ($\boldsymbol{wp_p} = \boldsymbol{wp_s}$)	0.152	0.243	0.387
Water quantity exchanged (Mm^3 / %)	381 (19.1%)	363 (27.3%)	259 (38.9%)
Profit losses without water bank (M€ / %)	54.9 (4.9%)	208.1 (18.6%)	422.4 (37.8%)
Profit losses with water bank (M€ / %)	19.3 (1.7%)	151.9 (13.6%)	369.6 (33.0%)
Net public revenue (M€)	0	0	0
Decrease in labor demand without water bank (AWU / %)[a]	488 (1.3%)	2,417 (6.5%)	5,202 (14.0%)
Decrease in labor demand with water bank (AWU / %)[a]	106 (0.3%)	751 (2.0%)	1,927 (5.2%)

[a] Labor demand measured in AWU (annual working units), which corresponds to the work performed by one person who is employed on an agricultural holding on a full-time basis.

Because of the differential behavior of farm types described above, water transfers have different effects on cropping patterns. Farm types with higher water productivity (mostly in C2 and C3 irrigated areas) can afford to purchase water from the bank, and during moderate water shortages, these farms do not need to modify their usual cropping patterns (those implemented under normal hydrological conditions); rather, changes in crop mixes occur only during severe and extreme droughts, these changes being more relevant as water scarcity became more acute. On the other hand, farm types with low-water-productivity crops (mainly in C5 and C1 irrigated areas) change their usual crop mixes to crops with lower water

[10] In the scenarios of extreme and severe droughts, a previously approved Drought Plan is implemented, assuring a minimum allocation of water for perennial crops in order to avoid the collapse of the orchards. However, it is worth pointing out that this allocation does not allow any profitable production. Thus, in drought situations, farmers with permanent crops are willing to participate in the bank purchasing water to obtain a complete harvest.

requirements in every simulated scenario, eventually switching to rain-fed crops in severe and extreme droughts.

Table 5.5 also shows the performance of the water bank in terms of economic (profit changes in the agricultural sector and public revenue) and social (changes in labor demand in the agricultural sector) impacts. When the water availability equals 75% of water rights, in the absence of a water bank, there are significant profit losses in the irrigation sector at the basin level, estimated to be 54.9 M€ (4.9% of the aggregated profit at full water availability). In contrast, the simulation results indicate that the creation of a water bank would reduce these losses to 19.3 M€ (only 1.7% of aggregated profit under normal hydrologic conditions). Thus, compared to the current situation (without a water bank), the operations of the water bank proposed in this drought scenario would increase economic efficiency at the basin level by 35.6 M€ (3.2% of normal aggregated profit). Clearly, the profit losses are larger, both in absolute and relative terms, when the drought is more severe. However, for severe and extreme drought episodes, the water bank considered here has a more relevant role as a water efficiency enhancer since the market operations implemented by the water bank could increase the aggregated profit by 56.2 M€ and 52.8 M€ (5.0% and 4.7% of normal aggregated profit), respectively. These results clearly demonstrate the potential of water banks to increase economic efficiency in the GRB.

Regarding economic impacts, it is clear that a water bank management strategy that aims to maximize the amount of water exchanged (i.e., $wp_p = wp_s$) involves a net public revenue of zero. Thus, considering that in a real-world setting, the transaction costs that are borne by the public water agency are not negligible, the implementation of this market strategy would lead to a hidden subsidy that is equal to all costs actually covered by the agency in order to create and manage the water bank. In the next section, this point will be further discussed.

Finally, with respect to social impacts, Table 5.5 also shows how the water bank led to a reduction in the aggregated losses of labor demand at the basin level for all drought scenarios considered. For example, when a severe drought is in place, the labor demand in the irrigation sector is reduced by 6.5% if no water market instrument is implemented. However, this loss can be reduced to only 2.0% if a water bank is considered an instrument for managing water scarcity because the water bank favors the maintenance of the most profitable crops, which also require the most labor. In fact, as seen in Table 5.5, the role of a water bank proposed as an economic instrument to mitigate the negative social impacts of water shortages is more relevant when the drought is more acute, both in absolute and relative terms. These results also indicate the potential of water banks to improve the management of drought periods in the GRB from a social perspective.

5.5.3. Performance water banks with operational costs

This section analyzes the simulated outcomes accounting for the water bank operation costs that are borne by the public agency that manages the water bank. This agency would thus implement a differential water price between water purchases and sales ($wp_p \leq wp_s$), generating a net positive public revenue that aims to cover these costs. Table 5.6 shows the results obtained for the different scenarios of reduced water availability (the coefficient of water availability δ equals to 0.75, 0.50 or 0.25) when there is a difference of €0.01/m^3, €0.02/m^3 and €0.03/m^3 between the purchase and sale price. Similar to the case in which water bank operation costs are assumed to be zero, the results measure the performance of the proposed water bank in terms of the clearing prices, volumes of water transferred, aggregated gross margin in agricultural sector, public revenues and total demand for labor.

The first result worth noting is that implementing a differential price of only €0.01/m^3 would be enough to cover all public transactions costs that are incurred from the creation and management of a water bank in the GRB. In fact, the net public revenues that would be generated would range from 3.8 M€ for the case of a moderate drought (δ=0.75) to 2.6 M€ for the case of an extreme drought (δ=0.25), far above the estimated costs (1-2 M€ annually). Therefore, this minimum differential price strategy would be enough for cost recovery, thus avoiding the need to subsidize this market institution.

Table 5.6. Market balance with water bank operation costs

		Coefficient of water availability		
		75%	*50%*	*25%*
Water quantity used (Mm³)		1,995	1,330	665
Differential cost €0.01/m³	Water purchase price (wp_P) (€/m³)	0.142	0.239	0.377
	Water sales price (wp_s) (€/m³)	0.152	0.249	0.387
	Water quantity exchanged (Mm³ / %)	381 (19.1%)	337 (25.4%)	259 (38.9%)
	Profit losses without water bank (M€ / %)	54.9 (4.9%)	208.1 (18.6%)	422.4 (37.8%)
	Profit losses with water bank (M€ / %)	23.1 (2.1%)	155.6 (13.9%)	372.1 (33.3%)
	Net public revenue (M€)	3.8	3.4	2.6
	Decrease in labor demand without water bank (AWU / %)	488 (1.3%)	2,417 (6.5%)	5,202 (14.0%)
	Decrease in labor demand with water bank (AWU / %)	105 (0.3%)	863 (2.3%)	1,927 (5.2%)
Differential cost €0.02/m³	Water purchase price (wp_P) (€/m³)	0.132	0.235	0.367
	Water sales price (wp_s) (€/m³)	0.152	0.255	0.387
	Water quantity exchanged (Mm³ / %)	381 (19.1%)	316 (23.8%)	235 (35.4%)
	Profit losses without water bank (M€ / %)	54.9 (4.9%)	208.1 (18.6%)	422.4 (37.8%)
	Profit losses with water bank (M€ / %)	26.9 (2.4%)	159.3 (14.2%)	374.7 (33.5%)
	Net public revenue (M€)	7.6	6.3	4.7
	Decrease in labor demand without water bank (AWU / %)	488 (1.3%)	2,417 (6.5%)	5,202 (14.0%)
	Decrease in labor demand with water bank (AWU / %)	104 (0.3%)	877 (2.4%)	1,905 (5.1%)
Differential cost €0.03/m³	Water purchase price (wp_P) (€/m³)	0.128	0.236	0.357
	Water sales price (wp_s) (€/m³)	0.158	0.266	0.387
	Water quantity exchanged (Mm³ / %)	254 (12.7%)	312 (23.5%)	235 (35.3%)
	Profit losses without water bank (M€ / %)	54.9 (4.9%)	208.1 (18.6%)	422.4 (37.8%)
	Profit losses with water bank (M€ / %)	30.0 (2.7%)	162.6 (14.5%)	377.0 (33.7%)
	Net public revenue (M€)	7.6	9.4	7.0
	Decrease in labor demand without water bank (AWU / %)	488 (1.3%)	2,417 (6.5%)	5,202 (14.0%)
	Decrease in labor demand with water bank (AWU / %)	110 (0.3%)	880 (2.4%)	1,904 (5.1%)

Focusing on the results for a differential price of €0.01/m³, the water quantities exchanged by the water bank are very similar to those reported in the case where water bank operation costs were zero. In fact, the quantities are almost the same for the moderate and extreme drought situations and are only 26 Mm^3 lower (8% less, from 363 to 337 Mm^3) for a severe drought. These findings are in line with those of Garrido (2000), who simulated a spot water market between four irrigation districts in the GRB and found that with operation costs of €0.006/m^3 and with a water availability of 50%, 18.2% of used water would be traded. However, this study shows that in case that these costs increase up to €0.018/m^3, market activity would be reduced to 15.3% over the region's total water availability.

These similar water exchanges involve similar economic and social performances. For example, when implementing a differential price of €0.01/m^3 in an extreme drought situation, the losses are 372.1 M€, only 2.6 M€ more than the performance achieved without implementing such a differential price. Moreover, this difference is equivalent to the net public revenue generated in this case. This result means that the economic inefficiency produced by implementing a differential price strategy is negligible at this level. Regarding social impacts, there is no difference between the results obtained in this simulated scenario and those when considering water bank operation costs of zero.

As shown in Table 5.6, only an insignificant economic inefficiency appears for a severe drought scenario. In this case, since the price of the water exchanged is slightly lower, the profit losses when water bank is operating are 155.6 M€, 3.7 M€ more than the performance achieved when no differential price is implemented. However, in this case, the public revenue is only 3.4 M€, meaning that recovering the water bank operation costs would lead to an economic inefficiency of 0.3 M€, which is less than 0.1% of the aggregated profit. Slightly more relevant is the change in social performance, since the decrease in aggregated labor demand in this case is 15% higher than the scenario in which no differential price strategy is implemented (increasing from 751 to 863 AWUs).

Although the scenarios considering differential price strategies of €0.02/m^3 and €0.03/m^3 are more than needed to recover costs, one point is worth noting. The relatively large increase in economic efficiency generated by the operation of the water bank would allow for the implementation of reasonable differential price strategies (i.e., less than €0.03/m^3) without significantly impacting market performance, both in economic and social terms. Thus, there is room to consider water banks not only as an instrument for enhancing economic efficiency regarding water use but also as a mechanism for collecting extra water fees in case this is considered necessary by decision makers.

5.6. Concluding remarks

Although the implementation of a publicly run active water bank is usually suggested as a demand-side policy instrument for reallocating water resources during drought periods, there is little evidence regarding their potential performance. This study attempts to fill this gap in empirical knowledge by simulating the implementation of this kind of water bank, taking into account different future scenarios of reduced water availability (cyclical scarcity) at the basin level. For this purpose, the bank proposed it is assumed to operate during drought periods, first implementing public offers to purchase temporary rights, and afterwards selling all the rights that were previously bought through public sale offerings. The key idea guiding this proactive strategy is that by fixing purchase and sale prices properly, this bank could maximize the quantity of water that is reallocated among irrigators, thus enhancing water use efficiency in a similar way to a spot water market or a passive water bank operating through a clearing house. Although the potential outcomes of all these water trading instruments could be similar in terms of increased water use efficiency, it is worth noting that the proposed active water bank has the advantages of involving lower operation costs and more effective control over market operations (reducing environmental and social negative externalities).

The analysis of this case study (Guadalquivir River Basin) yielded evidence that the implementation of a water bank could lead to large water transfers, potentially ranging from 19% of the total water used in the case of a moderate drought to almost 40% in case of an extreme drought. This traded water is accompanied by an increase in farmers' total profit (i.e., an increase in the economic efficiency) and in the aggregated agricultural labor demand (i.e., positive social impact enhancing rural development). Both of these positive impacts serve to minimize the effects of cyclical scarcity. In fact, the more severe the drought is, the greater the increase in profit and labor demand will be.

The outcomes obtained from the simulation of different water availability scenarios show that a difference in the purchase and sales prices of only €0.01/m^3 would be enough to cover all operational costs that are borne by the public agency that manages the bank. Moreover, the results demonstrate that a strategy that employs differential prices greater than €0.01/m^3 could generate an extra source of public revenue without significantly affecting bank performance in terms of economic efficiency or labor demand.

These results lead to the conclusion that a water bank that is designed to reallocate water resources between irrigators is a very useful instrument for coping with cyclical scarcity in closed basins, such as the GRB, since water trading would significantly mitigate the negative effects of droughts. In this sense, we suggest that policymakers pay attention to this economic instrument to improve water management and encourage the creation of drought water banks.

Nonetheless, it should be noted that water banks should not be considered panaceas for solving all management problems during drought periods. In fact, these tools should be just understood as complementary economic instruments and should be combined with other demand-side policy instruments. Moreover, the results reported in this paper must be understood as the maximum potential performance of the water bank proposed for implementation. The simulation model presented here ignores some possible barriers to trade that are difficult to model when using mathematical programming techniques, such as legal, administrative, cultural, psychological, technical and environmental barriers. In fact, although the proposed water trading instrument could minimize many of the barriers (all but technical and environmental ones), it should be noted that these limitations of the model could lead to overestimating the amount of water traded and the increase in economic efficiency. In this sense, further research is suggested to refine this kind of modeling approach, especially by considering: a) more realistic constraints related to physical connectivity among agents (also including groundwater users), b) transportation costs when transfers require specific infrastructures, and c) environmental constraints to maintain minimum environmental water flows as fixed in the Basin Management Plan.

In any case, all abovementioned limitations should not cloud the results that suggest the promising potential performance of the proposed active water plan as an instrument to enhance water use efficiency during drought periods. In this regard, further research is also suggested to analyze other alternative management strategies for active water banks, such as implementing auction systems for water bank purchases and sales in order to minimize private agents' surplus and maximize increases in efficiency. Similarly, implementing double-purpose purchases, both for reallocation (increasing efficiency in water use) and buyback (decreasing withdrawals to improve environment), would be worth exploring in order to analyze the trade-off between both objectives.

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Capítulo 6
Conclusions

After addressing the proposed research objectives, we are able to validate the initial research hypothesis: economic instruments for water management can help to foster better water governance in Spain by permitting a balanced trade-off between economic efficiency and environmental sustainability.

This research has highlighted the necessity of setting up demand-side management tools, such as economic instruments that provide the necessary flexibility in water rights systems, due to the difficulty of further supply increases. The current rights system in Spain is not flexible enough to respond to pressure from agriculture, population growth, the environment (ecological flows) and climate change. This lack of flexibility results in decreased water use efficiency because rights reallocation cannot allocate to those uses that are socially more in demand at a given moment. This context justifies the purpose of this research in attempting to analyze the potential impacts of two water economic instruments, *water pricing* and *water banks*, both aiming at a more efficient reallocation of water resources in irrigation districts located in a "closed basin". The empirical applications proposed are developed in the Guadalquivir River Basin (southern Spain), a basin currently closed to new users due to a significant increase in water demand over the past few decades and the impossibility to further increase the water supply at an affordable cost. This research has provided several contributions to filling the knowledge gap concerning these economic instruments by responding to the specific proposed **empirical** and **methodological** objectives.

6.1. Empirical conclusions

6.1.1. Implementing volumetric water pricing within the agricultural sector

First, we assessed the feasible impacts of water pricing in two different irrigation districts within the Guadalquivir River Basin. As demonstrated in the previous literature, both Chapter 2 and Chapter 3 have shown that water pricing in selected irrigation districts is not expected to reduce water use at low water prices. Indeed, the increase of water tariffs up to full-cost recovery levels (bellow €0.10/m^3) has not resulted in any significant change of farmers' cropping patterns towards those crops with lower water requirements, such as herbaceous crops or rain-fed crops. Thus, any increase in water pricing would result in a mere monetary transfer from farmers to the water agency.

Initial inelastic segments of the water demand curves are due to profitable crops (high value-added and usually high water consumption) that farmers are not willing to change in favor of other less profitable ones. Only very high water tariffs would move cropping patterns in the analyzed irrigation districts, resulting in a decrease of water use at the expense of significant profit and labor losses. Nonetheless, the length of the initial inelastic segment varies between irrigation districts and farm type. As we can see in Chapter 3, considering an irrigation district of average profitability for the empirical analysis, the inelastic segment is shorter (to €0.18/m^3) than in the highly profitable irrigation district considered in Chapter 2 as a case study (to €0.30/m^3).

It should be noted that by simulating the impact of water pricing using mathematical programming models we can also estimate a series of indicators of interest for policymakers, covering economic (gross margin) and social (agricultural labor demand) issues. In this sense, the results show a significant loss in agricultural income in both segments of the demand curve, inelastic and elastic. Moreover, in the elastic segment, these losses to farmers' incomes are higher than the gains achieved in public revenue, resulting in losses of economic efficiency. Furthermore, pricing irrigation water within in the elastic segment would involve significant losses in employment generation, thus jeopardizing rural development in the regions where irrigation districts are located.

Finally, based on the results above, we can conclude that water pricing in closed basins within the elastic segment has limited effects related to water use (i.e., it does not result in significant water savings since farmers are not induced to change their cropping plan). Thus, considering cost recovery tariffs (included in the inelastic segment), we can affirm that water pricing is neither a reasonable tool to reallocate water resources nor to improve efficiency in water uses.

6.1.2. Lesson learned from the international experience regarding water banks

An extensive review of the literature on water banks implemented to date throughout the world was undertaken in Chapter 4. This enabled us to verify that water banks can show different designs, with useful mechanisms to achieve different policy objectives. In this way, water banks facilitate water transfers towards uses with greater added-value (including environmental uses), reduce static transaction costs associated with administration, contracting, monitoring and enforcement of water exchanges, and thus support more active water markets able to increase economic efficiency. Water banks also centralize market operations, which allows the public water agency to properly control potential negative externalities of water trading and prevent harmful speculation. All these factors make water banks useful economic instruments to minimize the negative impacts of water scarcity, whether it be cyclical or permanent.

Nonetheless, the review has also exposed some shortcomings in the implementation of water banks around the world. Regarding environmental externalities, water banks, as a water trading instrument, can change the water flow regimes, negatively impacting on water-related ecosystems. Furthermore, we note the high costs of designing and implementing them (the so-called dynamic transaction costs). With this in mind, we suggest that all water banks should aim to be self-financed as promoted by the Water Framework Directive (WFD), where full-cost recovery is a key issue.

Different types of water banks have been identified based on i) the public or private nature of the responsible institution; ii) the type of rights exchanged (permanent, temporary or option contracts); iii) the potential purpose of the reallocation of the resource (from lower-value to high-value), for the environment and for managing risk related to water availability; and iv) the management strategy that can be passive or active (buying water rights based on budget). Different combinations of these policy design attributes lead to different kinds of water banks, each with its own particular pros and cons. The analysis of the features of the different designs of this economic instrument has revealed what kind of water bank would be potentially more suitable for water management in a closed basin such as the Guadalquivir River Basin, where one of the most challenging issues is minimizing the impact of drought episodes.

6.1.3. Implementation of an active water bank for temporal reallocation of water within the agricultural sector

After considering all types of water banks, we propose a publicly run active water bank operating at basin level designed to temporarily reallocate water resources between farmers as the best option to be implemented in the Guadalquivir River Basin in Chapter 5. The implementation of this kind of water bank has been simulated in order to assess its potential performance considering different scenarios of water availability (cyclical scarcity due to droughts) in the basin considered as a case study. This analysis has shown that the implementation of such water banks in closed basins could lead to large water transfers that are accompanied by increases in farmers' total profit and aggregated agricultural labor demand. Both positive impacts serve to minimize the effects of cyclical scarcity. Note, however, that the more severe the drought, the greater the relative increase in profit and labor demand must be with respect to total water use. Moreover, this type of water bank is confirmed to be suitable because it involves low operational costs and provides more effective control over market operations (reducing environmental and social negative externalities).

However, it must be noted that though some possible barriers to trading (legal, administrative, cultural, psychological, technical and environmental) have been ignored in the simulation models because they are difficult to address with mathematical programming techniques. Some of these barriers are actually minimized by the water bank in comparison to water markets, but others are not. Thus, this limitation of the model could lead to overestimating the amount of water traded and the increase in economic efficiency. As pointed out below, there is room for further research in order to more realistically model the effects of these possible barriers.

6.2. Methodological conclusions

6.2.1. Comparing current approaches for simulating farmers' behavior

Regarding the **specific methodological objectives** analyzed, we observe that mathematical programming models have proven to be the best option to simulate farmers' behavior compared to other options such as econometric models. In this way, three well-known methods of calibration of objective functions were compared in Chapter 2: i) profit maximization; ii) Positive Mathematical Programming (PMP) and iii) Additive Multi-Attribute Utility Functions (MAUFs) maximization based on Weighted Goal Programming (WGP). After running the calibration procedures, it is evident that PMP and WGP better reproduce the farmers' behavior because of their smoother and more credible shape, so they are considered more accurate than profit maximization.

6.2.2. Simulating farmers' behavior calibrating individual Cobb-Douglas multi-attribute utility functions

To provide more in-depth knowledge about simulating farmers' behavior by using non-linear MAUFs, a new methodological approach has been developed in Chapter 3, based on a Cobb-Douglas utility function that is more coherent with economic theory than the WGP (additive MAUF) approach commonly used since it assumes neither a constant marginal rate of substitution between attributes nor total compensation between them, ensuring diminishing returns and a global optimum. The proposed method has been empirically implemented to simulate farmers' behavior in a real case study. This approach for calibrating the objective function is compared with the approach based on an additive MAUF, confirming the advantages of the Cobb-Douglas MAUF approach.

The study implemented reached two main conclusions. First, this Cobb-Douglas MAUF approach can be easily implemented for simulation purposes in real settings and could be a useful procedure for *ex-ante* simulations of policy instruments for any type of future scenario. Second, the calibration implemented using this approach is more precise since the resulting water demand curve has a smoother shape than in the other approaches reviewed, and this shape is more plausible because farmers are expected to make marginal changes when facing marginal external shocks. Thus, this new calibration method can produce fruitful outcomes for policy analysis because it provides a better simulation of results than more traditional approaches.

6.3. Guidelines for improving Spanish water law

6.3.1. Water pricing

Article 9 of the Water Framework Directive requires Member States to recover all costs of water services in order to achieve the environmental objectives targeted by this European normative instrument (i.e., good ecological status of all water bodies). However, the WFD also includes some exceptions to cost recovery in case the water pricing instrument used for this purpose does not effectively reduce water use and involves severe negative economic (profitability) or social (local development) impacts. However, Spain neither complies with the full-cost recovery requirement nor explains the implementation of this exception to this WFD article. In addition, current tariffs imposed to irrigation water users are paid on a per irrigated hectare basis. Both facts discourage water-saving behaviors and jeopardize the sustainability of water services in closed basins. In this sense, two major normative reforms are needed to improve water management: a general increase in irrigation water tariffs and a change in the way of charging them to users in order to be calculated on a volumetric basis.

In any case, in order to implement the suggested reforms, the result obtained in this study should be taken into account. A volumetric cost-recovering water pricing would not achieve a reduction in irrigation water use due to the initial inelastic segment of the demand curve typical of closed basins. Thus, the implementation of the cost-recovery principle would have a negligible environmental impact on water bodies. To save water and improve the ecological status of water bodies, water tariffs should be well above cost-recovery levels, leading to severe negative impacts, both in economic (economic inefficiency) and social concerns (employment generation). These evidences would be used as the justification needed to consider a (partial) exception to the implementation of the cost-recovery principle.

In this context, a compromise solution between the current situation and full-cost recovery implementation is suggested as a more reasonable option. Thus, we recommend a moderate increase in irrigation water tariffs (between €0.02/m^3-€0.05/m^3) and to charge these tariffs to irrigators on a volumetric basis. These changes would not decrease water use in the irrigation sector but would improve water management in two ways. On the one hand, this increased volumetric water tariff would ensure the sustainability of public water services, and thus the viability of economic activities using this resource. On the other hand, this would discourage any inefficient water use and prevent new users from requesting water for low-efficiency uses.

6.3.2. Water banks

Although Law 46/1999 allows the creation of water banks in Spain, this economic instrument actually can only be established in "exceptional situations of water scarcity", namely, special drought situations or severe overexploitation of aquifers. To enhance the potential performance of water banks within the Spanish water management framework, we suggest that they should be established immediately and remain open permanently, not only in "exceptional situations". This would allow several objectives to be achieved. A permanent open bank would be useful to train the agents to operate in this market (enhancing water bank activity) and to show the real value of water (clearing price reached through the water bank) and how it varies over time depending on water scarcity. Both are key elements to implement a water bank successfully.

Additionally, the implementation of water banks also requires political will, providing water agencies with the technical and human resources needed to manage them successfully. Thus, an important initial investment is needed for water banks in order to cover all institutional transaction costs dealing with the design and establishment of this instrument. However, we also suggest that once the water banks are established, static or operational transactions costs (such as contracting, monitoring and detection, prosecution and enforcement, as well as additional costs of maintaining the infrastructure) should be borne by water users operating in this market by fixing a differential price in public offers to purchase water and public offers to sell temporary water rights.

Furthermore, Spanish water law should also be amended in other aspects regarding water banks in order to enhance its performance. In this sense, the following adjustments are proposed:

a) The amount of water transferred from a particular user should be controlled by the water bank in order to reduce negative environmental externalities. Thus, the total water rights transferred have to be lower than the volume of water actually consumed (water extracted from the source that does not return to water bodies) in previous years. In other words, water banks should avoid transferring the fraction of water corresponding to return flows to have a neutral impact on the hydrologic system.

b) New water users, such as farmers, industries, etc., without water rights should be allowed to participate in water banks to be more economically efficient, effectively promoting water transfers towards higher value uses.

c) The transparency of the market should also be improved, publishing (e.g., on a website) all information about the operations arranged (parties involved, prices and trading volumes agreed, terms of the offer, etc.).

d) It is necessary to establish a method to avoid fraud after the sale of the water rights; this is, to guarantee that the temporary water rights sold are not used afterwards by the seller.

Finally, it should be noted that water trading instruments (water banks or water markets) should not be considered *panaceas* for solving all management problems during droughts. In fact, these tools should be just understood as complementary economic instruments to be combined with other demand-side policy instruments.

6.4. Future research

This study has identified several issues for future research, regarding not only empirical research but also methodological research. Among **empirical research** topics regarding *water pricing*, it is suggested to analyze the impact of water pricing in the long run, where the possibility of structural changes in irrigated farms such as farm size or changes in the irrigation techniques could be considered. Moreover, other water pricing alternatives in addition to the volumetric one should be analyzed in the future, such as a block-rate system (incremental price per fixed volumes consumed). Finally, more empirical evidence regarding the shape of water demand curve would be useful for policy decision making. More concretely, the range of inelastic segments could be analyzed at the farmer level, enabling evaluation of the heterogeneous impact of water pricing among farmers. This would allow a more sensible policy and decision making in the implementation of water pricing, justifying the partial exemption of water pricing when required based on objective economic, environmental and social criteria.

Regarding *water banks*, other alternative managements could be analyzed, such as implementing auction systems for water bank purchases and sales in order to minimize private agents' surplus and maximize efficiency in water use. Furthermore, a double-purpose water bank, both for reallocation (increasing efficiency in water use) and buyback (decreasing withdrawals to improve environment), would be worth exploring in order to analyze the potential impacts of this design of water bank and the trade-off between both objectives.

As explained in the thesis, the simulation of barriers hindering the performance of water banks is difficult to model with mathematical programming techniques. Thus, a way of simulating these barriers needs to be developed in order to achieve a more realistic simulation of the performance of this policy instrument. Thus, further research is also needed in this sense.

Regarding **methodological research,** the new non-interactive method to elicit Cobb-Douglas MAUFs would need to be implemented in other agricultural systems and for the analysis of other policy instruments (e.g., agricultural subsidies, insurance schemes, etc.) to confirm that the performance of the simulated farmer' behavior is sufficiently accurate. Indeed, some others functional forms of the utility function could be elicited and tested, such as the constant elasticity of substitution function (CES, a more general form than the Cobb-Douglas function). Moreover, it would be worthwhile implementing experiments to test that MAUF parameters remain constant over time (by using multiple elicitation procedures with the same decision-makers in different time periods).