Departamento de Estructura Económica

Essays in Applied Microeconomics on Water Resources Management

DOCTORAL DISSERTATION

Author:

Marta Suárez-Varela Maciá

Supervisors:

Andrés J. Picazo Tadeo

Juan A. Mañez Castillejo

Doctorado en Economía Industrial

Octubre 2017

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Agradecimientos

Quisiera aprovechar estas primeras líneas para expresar mi gratitud a todas aquellas personas e instituciones que, de un modo u otro, han contribuido al desarrollo de esta tesis doctoral.

En primer lugar, quiero agradecer a los doctores Andrés J. Picazo Tadeo y Juan A. Máñez Castillejo, mis directores de tesis, por la dedicación y el celo con el que me han orientado y realizado las sucesivas correcciones que han dado lugar a este trabajo.

En particular, le agradezco a Andrés J. Picazo la confianza que depositó en mí al acogerme, hace ya cuatro años, en el proyecto ECO2012-32189 del que es investigador principal. Fue de su mano que llegué a la Universidad de Valencia, dándome con ello la oportunidad de iniciar una carrera académica. Desde entonces, no puedo menos que estar enormemente agradecida por su amistad, su sincera implicación y la manera en la que me ha alentado y guiado a lo largo de este proceso, apoyándome en cada uno de mis pasos y decisiones. Su contribución, además de quedar plasmada en las hojas de esta tesis doctoral, ha sido sin duda esencial no solo para mi formación como doctora y docente, sino para mi desarrollo personal. Muy especialmente agradezco su empeño en inculcarme el valor del rigor científico y la importancia de corresponder adecuadamente a la responsabilidad que la sociedad deposita en nosotros como docentes.

Durante el periodo predoctoral, he tenido así mismo el placer de trabajar con excelentes investigadores de otras universidades -compañeros y amigos- que han contribuido de forma muy sustancial al desarrollo de este trabajo. En primer lugar, expresar mi gratitud a Roberto Martínez Espiñeira por hacerme sentir como en casa durante mi estancia en la Memorial University of Newfoundland (Canadá) en 2015. Su constante disponibilidad, su paciencia para transmitirme algunos de los métodos econométricos que posteriormente utilicé en esta tesis, y aquellas horas frente al ordenador, contribuyeron de forma sustancial al desarrollo de la misma. Likewise, I would like to thank Ariel Dinar for its support, kindness and dedication during my stay in the School of Public Policy in UC Riverside (USA) in 2016 and beyond. His collaboration is indeed very much appreciated. A Marian García Valiñas debo agradecer entre otras cosas las frecuentes charlas y discusiones académicas que encauzaron algunos de los trabajos que hoy la componen, así como todos los consejos que me ha brindado con sincero cariño y generosidad. Finalmente, mostrar mi agradecimiento a mis compañeros de la Universidad de Granada, en especial a Jorge Guardiola Wanden-Berghe y Francisco González Gómez, que acompañaron mis primeros pasos en investigación, por la confianza que depositaron en mí y por haberme animado a seguir este camino.

De igual modo, quisiera expresar mi gratitud a los compañeros del Departamento de Estructura Económica de la Universidad de Valencia no solo por el apoyo recibido en mi proceso de formación, sino también por la cálida acogida y sincero compañerismo que han hecho mi día a día más grato y llevadero. En particular, a José Antonio Martínez Serrano, por su sincera preocupación y sus, aunque a veces incómodos, siempre sabios consejos; y a Mercedes Beltrán, por su amistad, su empatía y las innumerables veces que me ha ofrecido su cariño y apoyo.

Me llevo asimismo de esta experiencia grandes amigos y “compañeros de fatigas” a los que no quiero dejar de agradecer los buenos momentos compartidos. A los compañeros del doctorado: Adri, Adriana, Carlos, Danny, Eli, Ernesto, Fede, Jesús, Jorge, Laura, Luis, Mariola, Marta, Paula y Rubén; a los del Departamento de Contabilidad, Amine y Osama; y a otros, como Bea. En especial quisiera dar las gracias a Carlos, Danny, Fede y Jorge por su amistad y las incontables ocasiones en que me han permitido, cuando las crisis doctorales hacían sus estragos, invadir sus despachos “sólo para molestar”. Esos momentos de escucha resultaban siempre imprescindibles para superarlas.

A mis amigas de Granada, con cuyo apoyo he tenido la suerte de contar en éste, al igual que en otros muchos momentos importantes de mi vida en que lo he necesitado (Ana, Aida, Carolina, Elo, Isa, Lola, Mariló, Marina y Rosa): gracias por

estar tan presentes, incluso en la distancia. A los amigos que me habéis acompañado en esta nueva etapa que comencé en Valencia (Andrea, Anna, Ana Mar, Carlos, Danny, Elisa y Jorge): gracias por llenarla de afecto y momentos inolvidables. También vuestra amistad ha trascendido en mucho el ámbito de esta tesis doctoral.

Por último, pero no por ello menos importante, quisiera agradecer a mi familia por apoyarme en todo momento. En primer lugar, a mis padres y a Joaquina, porque sin su esfuerzo por ofrecernos una atmósfera en la que pudiéramos crecer con seguridad y amor, ni mi hermano ni yo seríamos las personas que somos hoy. Por mostrarme, además, con su ejemplo, que el empeño y la dedicación terminan dando sus frutos. A Pepe, a quien tengo la fortuna de tener por hermano, gracias por acompañarme y apoyarme en éste y otros muchos aspectos de mi vida. En definitiva, sin vuestro cariño y guía constante esta tesis no hubiera sido posible.

Para finalizar, he de agradecer al Ministerio de Economía y Competitividad de España y, en particular al proyecto ECO2012-32189 por financiar la realización de esta tesis doctoral a través de un contrato predoctoral para la Formación del Personal Investigador (FPI) durante el periodo 2014-2018 (Referencia BES-2013-062824).

A todos, gracias.

Introduction

Water is both an essential good and a scarce resource that exhibits several traits that have attracted the attention of economists. On the one hand, it is a merit good that contributes to meeting economic, environmental and social goals and generates substantial externalities. On the other hand, it satisfies a basic human need, so universal access and affordability should be guaranteed. As an economic good, an optimal allocation of the resource between competing uses such as industry, agriculture, households and ecosystems must be ensured (OECD, 2003). In the scientific sphere, growing interest in water-related issues has prompted the emergence of several lines of research in the field of economics that aim to contribute to the improvement of water resources management. Broadly, these lines can be classified into two categories: agricultural water economics and urban water economics, with the latter including residential and industrial uses.

In the residential sector, water management is expected to face several challenges. In the forthcoming decades, the global population is predicted to undergo exponential growth, going from an estimated 6 billion people at the beginning of the 21st century to 9 billion by 2050 (UN Population Division, 2015). In order to satisfy the growing needs of the population in terms of food production, energy generation, industry and residential demand, an unprecedented growth in water demand is forecast (World Bank Group, 2016). Moreover, climate change will reduce the availability of freshwater resources, making supply more unpredictable and volatile (IPCC 2015). Thus, with more than half of the global population living in regions suffering from at least moderate water stress by 2050 (Arnell, 1999, 2000;

Schlosser et al., 2014), water scarcity will likely pose a substantial risk for humankind in the near future (World Economic Forum, 2017).

Likewise, urbanization trends may represent an additional pressure on urban water management (Biswas and Tortajada, 2009). It is predicted that by 2050 around two-thirds of the world's population will reside in urban areas. Moreover, the trend towards ever-bigger urban agglomerations also entails a potential threat and increased complexity for urban water services (UN Population Division, 2014; Biswas and Tortajada, 2009). In fact, the number of cities that are home to over 10 million inhabitants - the so-called "mega-cities"- has almost tripled just since 1990; and by 2030, 41 urban centres are predicted to exceed that size (UNESCO, 2016).

As a consequence, water management will have to cope with major challenges in the years to come, thus making the efficient and sustainable use of water resources a priority for governments and supranational institutions.

Residential water management policies involve both supply and demand interventions. On the supply side, solutions often entail the construction of new infrastructure such as canals or desalination and wastewater recycling plants. However, traditional supply interventions have usually proved costly both in economic and ecological terms. Moreover, they have sometimes been found to generate a higher level of dependence in arid regions, as demand usually adapts to the increase in availability (World Bank Group, 2016).

Consequently, demand-side policies have proved to be more effective. Policy options in this respect can be divided into two main categories: pricing and non-pricing approaches. Pricing incentives mainly involve raising prices and designing appropriate pricing schemes that promote an efficient use of the resource (Olmstead and Stavins, 2009). Particularly, tariff schemes in the form of increasing block rates are widely applied to achieve the simultaneous objectives of economic efficiency, environmental sustainability and social equity and affordability in residential water management (Olmstead et al., 2007; García-Rubio et al., 2015). Non-pricing policies focus on, among other issues, behavioural interventions such as educational campaigns, rebate programmes for the installation of water-efficient technologies or temporary service interruptions.

In the same vein, along with demand and supply interventions, making urban water services management more efficient and climate-resilient proves essential in order to ensure supply continuity under the increasing pressures created by a growing population, climate change and rising income worldwide.

Within this framework, this doctoral dissertation aims to address a number of issues related to water resources management and the provision of the service at a residential level. First, we analyse residential water demand policies. Second, we examine the design of appropriate pricing schemes and taxes that help improve allocation and simultaneously achieve the objectives of sustainability, equality and an efficient use of the resource. In third place, we address the effect of environmental attitudes as well as pricing and non-pricing policies on certain averting behaviours related to water consumption. Lastly, we cover the measurement and improvement of efficiency in residential water management.

The four essays included in this dissertation share the common feature of using Spanish data for empirical purposes. However, it should be noted that the conclusions and policy implications are intended to be of general interest for a wide variety of settings and institutional backgrounds. The following paragraphs briefly describe the Spanish water sector, paying special attention to the main characteristics of its institutional and regulatory framework.

Spain is one of the highest-ranking European Union countries in the water exploitation index[1] (EEA, 2012), and most of its territory is subject to water stress or severe water stress. Moreover, precipitations are expected to decrease (Jacob et al., 2014) and droughts to become more prevalent in the near future (OECD 2011). Nevertheless, it is one of the countries with the highest per capita consumption (Suárez-Varela et al., 2015).

Furthermore, it should be noted that in Spain, there is currently little possibility of increasing the availability of freshwater by building new hydraulic infrastructures (OECD, 2011). Thus, demand-side policies have taken a prominent role in the past few decades. In this regard, the EU Water Framework Directive (WFD henceforth), which came into effect in 2000, highlights the importance of economic and pricing tools as a means to achieve an efficient allocation of water resources. Since then, pricing has thus been the focus of particular attention.

In Spain, town councils (municipal governments) are responsible for urban water management, i.e. they must organize the provision of the service. Town councils must choose between in-house management or outsourcing water management. If they opt for outsourcing, local governments may choose to do so with a publicly-owned company, to a private company or to a public-private partnership (see García-Valiñas et al., 2013, for further information on the legal framework governing outsourcing). It should be noted that only the management can be privatized, as the assets and infrastructures must remain public property.

[1] Ratio of total water abstraction per year to total long-term renewable resources

Furthermore, in the absence of a legal framework or formal regulatory body that establishes certain criteria and common guidelines for urban water tariffs, town councils are also ultimately in charge of price-setting decisions, control and monitoring (García-Valiñas et al. 2013).

As a consequence, there is an extraordinary range of tariff systems (González-Gómez et al., 2012), with two-part tariffs (i.e., including both a fixed and a variable component) and variable structures in increasing blocks being the most common (AEAS-AGA 2013; AEAS, 2013). Despite the efforts made in recent years (European Environmental Agency, 2013), water prices in Spain still do not reflect either the full cost of providing the service or the associated environmental and resource costs (AEAS-AGA, 2013; García-Rubio et al., 2015), as would be expected following WFD implementation. Moreover, the fact that water distribution networks currently suffer from substantial underinvestment, are old and in poor conditions, with non-revenue water exceeding 25% (García-Rubio et al. 2015) may represent an additional pressure to gradually raise prices over time. These features, along with the particular institutional background and regulatory framework that will be discussed in more detail below, make Spain an interesting setting for the study of the issues addressed in this doctoral thesis.

In what follows, we present a brief outline of the four chapters of this PhD dissertation, devoting special attention to the aim and contributions of each.

Essay 1: Estimation of residential water demand. Accounting for substitution patterns and non-linear income effects'.

The first essay of this doctoral dissertation focuses on the modelling of residential water demand. As mentioned at the beginning of this introduction, pricing policies are expected to play a prominent role in dealing with the challenges posed by greater water scarcity in the near future. In this respect, the success of pricing policies depends on precise estimates of demand patterns and elasticities. In the residential water demand literature, most estimations of income and price elasticities rely on single-equation models that are implicitly based on assumptions of weak separability of water consumption – i.e. residential water demand does not depend on the price of other goods consumed within the household - and linear income effects – i.e. an increase in income has the same effect on water consumption independently of the part of the income distribution to which the household belongs.

Recognizing those limitations, in this paper we propose a more flexible system of demand estimation, the Quadratic Almost Ideal Demand System, or QUAIDS (Banks et al. 1997). Among other unique features, it offers two main advantages over previously-used specifications. First, it estimates demand as a system of equations, thus accounting for possible substitution and complementarity patterns and allowing the estimation of cross-price elasticities. Second, quadratic terms in the logarithm of expenditure are included, thus taking into account the likely curvature of the Engel curves for the goods considered in the system. Moreover, it satisfies the fundamental principles of demand – i.e. adding up, integrability. Consequently, it is expected to offer richer and more accurate estimates of demand patterns, as well as enabling the analysis of the welfare effects of certain public policies related to price and taxes.

Our analysis is implemented using the Spanish Consumer Expenditure Surveys for the period 2006-2012, a rotating panel of 151,000 households (between 19,000 and 23,000 per year), which constitutes a representative sample of the Spanish population and contains information on household consumption of up to 489 goods. Information on prices is taken from the Consumer Price Index statistics, while the average price for water is derived from the sample. For modelling purposes, we assume households follow a two-stage budgeting procedure in which they first decide how to allocate their total expenditure among durable and non-durable goods and then allocate their total expenditure on non-durables among several composite commodities – food and beverages, water, energy, households expenses other than water and energy, and all other goods. Estimation is performed through an iterative linear least-squares estimator (ILLE) proposed by Blundell and Robin (1999) and the likely endogeneity of total expenditure and the price for water is dealt with using a non-linear instrumental variable approach also proposed by Blundell and Robin (1999) and the augmented regression techniques of Hausman (1978) and Holly and Sargan (1982).

Our results are as follows: We obtain long-run price and income elasticities of -0.974 and 0.108 respectively, implying that water is a normal good, a necessity and relatively price inelastic. Moreover, our sample showed the existence of non-linearities in Engel curves for residential water demand as well as substitutability and complementarity with other groups of commodities. Conversely, water demand was not found to have good substitutes among non-durable commodities. In this context, most commonly-used functional specifications – i.e. linear, log-linear and double-log - seem to impose rigid restrictions on demand estimation, whereas QUAIDS is expected to be more consistent with consumer behaviour. In a last stage, we compare QUAIDS with the abovementioned common specifications. The evidence seems to suggest that the predictions of demand yielded by QUAIDS are both closer to the observed data than the other specifications and better fit the asymmetry of the real distribution for water demand.

Therefore, water demand should be estimated in conjunction with the rest of the goods consumed in the household and more theory-compatible models should be employed. Moreover, the model proposed in this paper could be used jointly with representative Household Budget Surveys that are available in many countries, in order to obtain regional and national predictions necessary to inform policy decisions.

Essay 2: A proposal for the analysis of price escalation in water tariffs. The impact of the Water Framework Directive in Spain

In order to achieve the simultaneous objectives of efficiency, equality, sustainability and affordability, tariffs in the form of increasing block rates are becoming more widespread (Olmstead et al. 2007). These tariff schemes implicitly introduce a certain level of escalation or progressivity in the price for water –i.e. the price per cubic metre rises as consumption increases. However, despite its extensive use, no metric has been developed to measure the degree of escalation in water tariffs. In the second chapter of this dissertation, we propose a metric that aims to measure the level of progressivity embedded in water tariffs at the level of the water service management unit (in our study, the municipality), independently of the specific tariff scheme applied (i.e. regardless of the presence of free-allowances, the number and size of the price blocks, and the existence and size of the fixed component), and that allows full comparability among municipalities.

Specifically, we propose two indicators: one that accounts for the influence of the fixed component of the tariff; and another that only accounts for the variable price structure. Moreover, in order to illustrate the usefulness of the proposed measures, we analyse the evolution of Spanish tariffs following the implementation of the WFD as well as the political, business and environmental factors influencing that evolution. To that end, we use a sample of 952 municipalities in Spain for 2000 and 2014, that is, before and after the entry into force of the WFD. In Spain, in order to properly implement the WFD, the main advisory bodies recommended that municipalities should change from previously existing tariff schemes – e.g. flat or volumetric rates - to tariffs in the form of increasing block rates (IBRs), and should increase the level of escalation in the tariffs (AEAS, 2014). Thus, an increase in the level of escalation should have been expected.

Our results seem to suggest that despite the formal efforts made by local governments to increase price escalation by implementing steeper price increases for higher consumption levels, the main price hikes were applied to the consumption ranges with a smaller share of consumers. We also find that, when only the variable component is accounted for, although tariffs are on average progressive before and after the implementation of the WFD, there has been a decrease in the level of price escalation during that period, contrary to what would have been expected as a result of the WFD implementation. However, when the analysis also takes the fixed component into account, tariffs are found to be regressive on average, albeit less so in 2014 than in 2000. Besides, we find that some factors related to water stress, ideological factors, socioeconomic characteristics, tourist activity and the ownership of the management company may affect the probability of adopting more progressive tariffs.

These summary metrics are intended to assist policy makers in conducting sound policy analysis when assessing efforts made by countries and/or municipalities to increase the price escalation of tariffs or performing comparative analysis between jurisdictions.

Essay Chapter 3: The role of environmental attitudes in averting behaviour with negative environmental externalities: A double-hurdle approach to bottled water demand

The motivation for the third chapter lies in the fact that urban water management policies may also have an influence on other decisions taken within the household. For instance, poor quality tap water, as well as some other factors related to the management of urban water services, may prompt households to adopt certain averting behaviours such as installing water filters or consuming bottled water. In this context, our objective in this chapter is threefold. First, because consuming bottled water is an averting behaviour that poses significant negative environmental externalities and can be substituted by other more environmentally-friendly alternatives (e.g. filtering water), we investigate the role of environmental attitudes and behaviours in the decision to adopt averting behaviours that create negative environmental externalities. Moreover, we distinguish between the two categories of pro-environmental actions identified in the literature according to the level of individual effort required: one-shot or efficiency behaviours –e.g. installing certain resource-saving devices- and curtailment behaviours –i.e. daily habits or sacrifices to save resources. Second, given that bottled water and tap water could be either complementary or substitutes, we aim to explore the possible impact of several pricing and non-pricing policies applied in many urban centres around the world regarding bottled water consumption. In third place, we propose an empirical strategy for modelling averting behaviours that addresses the problems stemming from a large percentage of zero consumption records.

For this purpose, we use a representative sample of 592 households from the towns of Baza and Guadix located in the southern Spanish province of Granada, from the year 2014. Spain is the ninth largest per capita consumer of bottled water in the world and the fourth among European countries. Moreover, as Baza and Guadix are in an area suffering from severe water stress (European Environmental Agency, 2012), numerous water conservation policies are being applied in the region, making it an interesting setting for this work. A double-hurdle approach is proposed to cope with the large percentage of zero consumption records. This allows us to model averting behaviours without departing from any particular hypothesis regarding the reasons why households do not adopt said behaviours –i.e. non-participation vs. corner solutions- and to test the underlying distributional assumptions in order to choose among specifications.

Our results seem to suggest that commonly implemented pricing and non-pricing policies for managing water demand, such as increases in tap water price and supply cuts could result in an increased demand for bottled water, thus leading to unintended consequences in terms of negative environmental impacts. We also find that fostering pro-environmental habits could prove very successful in limiting averting behaviours that create negative environmental externalities. Additionally, we find that failing to properly address problems stemming from the large percentage of zero consumption records when modelling averting behaviours could give rise to misleading conclusions.

Essay 4: *Ownership and performance in water services revisited: Does private management really outperform public?*

The last essay in this dissertation aims to revisit the relationship between ownership and performance in water utilities. Although there has been intense debate ever since the 70s on which form of ownership of urban water services management - public or private - is more efficient, after more than three decades the literature still remains inconclusive. In this study, our aim is to shed some light on the ongoing debate by using a novel empirical strategy based on Data Envelopment Analysis (DEA), which combines two methodological approaches: directional distance functions and metafrontiers.

Our contribution is twofold. On the one hand, using metafrontiers allows us to discern whether differences in performance between private and public decision-making units are due to either different capabilities of the managers -managerial efficiency- or the different production technologies available to public and private utilities - ownership efficiency. This is an important distinction, as there are reasons to believe that public and private units face different technological restrictions - i.e. legal and institutional frameworks. On the other hand, the use of directional distance functions offers the advantage of enabling a performance evaluation at the level of the different inputs involved in the production process. Since technological restrictions may affect the performance of particular production factors differently, the combination of these two approaches has the potential to provide new insights in comparison to previous studies that have used either metafrontiers or directional distance functions separately.

The analysis focuses on a sample of 70 Spanish municipalities of under 50,000 inhabitants with data from the year 2013. In 37 of these municipalities, provision of urban water services is managed in-house – by the city council itself- or has been outsourced to a public company. In the remaining 33 cases, the service has been outsourced either to a private company or a public-private partnership (PPP). Performance is assessed through the concept of technical efficiency, with the production process characterized by two outputs and three inputs. The two outputs are water delivered and population served, while of the three inputs, one is fixed – the length of the delivery network - and two are variable - labour and operational costs. The analysis is input-oriented, that is, it evaluates the ability to reduce inputs while maintaining the same level of outputs.

Our results are as follows: In the conventional scenario in which efficiency is assessed in a direction that reduces all variable inputs proportionally - radial efficiency- no significant difference is found between public and private operators, in line with most studies in the field. However, our combined methodological approach offers a rather different picture. On the one hand, mainly because of a technological advantage, private operators are found to be superior in the management of labour input. Nevertheless, public management units are shown to be more efficient in the management of operational costs. These results can be explained by the fact that public managers face more restrictive labour regulations and higher levels of absenteeism (Meier and O'Toole 2011), as well as by other institutional and political factors specifically related to this type of ownership. Conversely, private operators may face greater difficulties in managing operational costs as they have usually been found to operate in more complex environments (González-Gómez et al., 2011). Furthermore, public utilities are able to share some of the operational costs with the rest of the services provided by the same public administration, especially under in-house public provision.

In summary, our results suggest that our approach proves successful in uncovering some insights that were hidden to more conventional measures of performance. This has important policy implications, as it can provide a useful tool for a number of relevant stakeholders –from managers of utilities, to regulatory bodies and policymakers–, enabling them to make more informed decisions on which practices and regulations benefit the industry. Moreover, it contributes to the abovementioned debate on the relationship between ownership and efficiency in urban water services.

Introducción

Como bien esencial y recurso escaso, el agua ha gozado habitualmente de especial atención por parte de numerosos sectores de la sociedad. Múltiples son las características que concurren en este hecho. De un lado, el agua es un bien de mérito que contribuye simultáneamente a la consecución de objetivos económicos, medioambientales y sociales, y que, además, genera considerables externalidades. De otro, el agua satisface una necesidad humana básica, de forma que se debe garantizar el acceso universal a precios asequibles para la población. Asimismo, como bien económico que es, es necesario asegurar una distribución óptima de este recurso entre usos rivales: industria, agricultura, hogares y ecosistemas (OCDE, 2003).

La consecución de los múltiples objetivos a los que debe obedecer la gestión de este recurso demanda necesariamente la implementación de políticas públicas ambiciosas, que permitan alcanzarlos de forma simultánea. En su condición de ciencia social, la Economía no ha permanecido ajena a este fenómeno. Particularmente en el ámbito científico, el creciente interés en el estudio de los problemas relacionados con la gestión del agua ha motivado la aparición de diversas líneas de investigación en Economía. Estas líneas, que se enmarcan dentro de la Economía del Medioambiente y Recursos Naturales y, más específicamente, dentro de la Economía del Agua, pueden clasificarse en dos categorías principales dependiendo de los sectores a los que dirigen su atención. Estas son: Economía del agua agraria, y economía del agua urbana, incluyendo esta última tanto usos residenciales como industriales.

En el caso del sector residencial, centro de interés en esta disertación, las tendencias que afectan a su evolución son de muy diversa índole y plantean importantes retos a largo plazo. De un lado, las predicciones indican que la población mundial experimentará un crecimiento exponencial en las próximas décadas, pasando de los 6.000 millones de personas a principios de siglo XXI hasta los 9.000 millones hacia el año 2050 (UN Population Division, 2015). Satisfacer las crecientes necesidades de dicha población, tanto en términos de demanda de alimentos como de actividades industriales, generación de energía y demanda residencial, implicará un incremento sin precedentes en la utilización de recursos hídricos (World Bank Group, 2016). De forma paralela, se espera que el cambio climático reduzca la disponibilidad de agua dulce, convirtiendo su suministro en más impredecible y volátil (IPCC, 2014). Como consecuencia, a mediados del siglo XXI más de la mitad de la población residirá, previsiblemente, en regiones con estrés hídrico moderado o severo (Arnell, 1999, 2000; Schlosser et al., 2014). Así pues, la escasez de agua constituirá uno de los principales riesgos para la humanidad en el futuro (World Economic Forum, 2017).

Las tendencias de desarrollo urbano pueden representar una presión adicional para la gestión urbana del agua (Biswas and Tortajada, 2009) dado que se espera que, en torno a 2050, aproximadamente dos tercios de la población resida en zonas urbanas. Además, la propensión a la formación de aglomeraciones urbanas de mayor tamaño implicará una creciente complejidad en la gestión de los servicios urbanos de agua (UN Population Division, 2014; Biswas and Tortajada, 2009). De hecho, desde 1990 el número de ciudades con más de 10 millones de habitantes - las denominadas “megaciudades” - casi se ha triplicado, y para el 2030 se prevé que 41 centros urbanos superen dicho tamaño (UNESCO, 2016). Como consecuencia de las tendencias señaladas, cabe esperar que la gestión de servicio urbano de agua afronte grandes desafíos en los próximos años. Así pues, la gestión eficiente y sostenible de este servicio debe ser un objetivo prioritario para gobiernos e instituciones supranacionales.

En términos generales, las soluciones de política pública destinadas a evitar o paliar los efectos de estas previsiones a nivel residencial pueden agruparse en políticas de demanda y políticas de oferta. Por el lado de la oferta, las intervenciones han pasado tradicionalmente por la construcción de nuevas infraestructuras, tales como embalses, desalinizadoras o plantas de reciclaje de aguas residuales, con el fin de incrementar la disponibilidad del recurso. Sin embargo, estas soluciones de oferta se han revelado habitualmente excesivamente costosas, tanto desde un punto de vista económico como ecológico. Además, se ha constatado que, en regiones áridas, éstas terminan derivando en una mayor dependencia de este recurso, dado que la demanda tiende a ajustarse a la mayor disponibilidad del mismo (World Bank Group, 2016).

En consecuencia, las políticas de demanda han adquirido en las últimas décadas una creciente popularidad. Entre las categorías incluidas en las opciones de política pública por el lado de la demanda se encuentran principalmente dos: políticas tarifarias, y el resto de políticas -denominadas genéricamente como no tarifarias. Por un lado, los incentivos tarifarios se dirigen principalmente a incrementar el precio del agua y, también, al diseño de esquemas de precios que promuevan un uso eficiente del recurso (Olmstead and Stavins, 2009). En particular, las estructuras de precios en forma de bloques crecientes, que permiten alcanzar simultáneamente los objetivos de eficiencia económica, sostenibilidad medioambiental y equidad social, son actualmente las más recomendadas (Olmstead et al. 2007; García-Rubio et al. 2015). Las políticas no tarifarias, por su parte, a menudo se concentran en intervenciones que persiguen modificar la conducta de los usuarios del servicio. Entre ellas cabe citar las campañas educativas, los descuentos en la adquisición de dispositivos ahorradores de agua o las interrupciones temporales del servicio.

Una tercera vía de adaptación a las previsiones demográficas y medioambientales observadas, son las políticas destinadas a mejorar la provisión del servicio urbano de aguas. Ante las crecientes presiones derivadas del cambio climático y la concentración de la población en centros urbanos, la continuidad en el suministro de agua pasa necesariamente por una gestión más eficiente del servicio y una adecuada financiación de las redes e infraestructuras. A fin de conseguir este objetivo, resulta imperativa la implementación de políticas que propicien sustanciales mejoras en la gobernanza del servicio urbano de aguas.

En este contexto, la presente tesis doctoral tiene por objeto abordar una serie de aspectos relacionados con la gestión de los recursos hídricos y la provisión del servicio a nivel residencial. En primer lugar, se analiza el efecto y adecuación de las políticas de demanda de agua anteriormente comentadas. En segundo lugar, se estudia la influencia de factores medioambientales, políticos y económicos en los procesos de fijación de precios y gobernanza en la provisión del servicio. Por último, se aborda la problemática relativa a la valoración y mejora de la eficiencia en la gestión del servicio urbano de agua.

Con este propósito, la presente tesis se compone de cuatro capítulos que comparten un mismo contexto geográfico e institucional para fines empíricos: la prestación del servicio urbano de agua en España. No obstante, las conclusiones e implicaciones de política pública pretenden ser de interés general para una gran variedad de entornos y contextos institucionales.

Con uno de los índices de explotación hídrica[2] más elevados de la Unión Europea (EEA, 2012), España se encuentra actualmente sometida a un estrés hídrico elevado o severo en la mayor parte de su territorio. Además, se espera que en el futuro se produzca una reducción sustancial en el volumen de precipitaciones (Jacob et al. 2014), con el consiguiente aumento en la prevalencia de sequías (OCDE, 2011). Sin embargo, contrariamente a lo que cabría esperar, en lo que a demanda de agua para uso residencial se refiere, España sigue siendo uno de los países europeos con mayor consumo per cápita (Suárez-Varela et al. 2015). A estas circunstancias ha de sumarse que el potencial para incrementar la disponibilidad de agua dulce mediante la construcción de nuevas infraestructuras hidráulicas se encuentra actualmente muy limitado (OCDE, 2011). Por tanto, las políticas de demanda han adquirido un papel destacado en los últimos años. Entre ellas, han sido en particular las políticas de precios las que han recibido especial atención. Esto se debe en gran medida a que la Directiva Marco del Agua de la Unión Europea (DMA), que entró en vigor en el año 2000, otorga una vital importancia a las políticas económicas y tarifarias como medio para alcanzar un reparto eficiente de los recursos hídricos.

En lo que respecta a la gobernanza en la gestión urbana del agua, cabe destacar que el nivel de administración responsable de la provisión del servicio es el municipio, si bien se contemplan diversas fórmulas legales de gestión. En primer lugar, el municipio puede elegir gestionar el servicio dentro del propio ayuntamiento, o externalizarlo. En el segundo caso, el gobierno local puede optar por utilizar una empresa pública, una privada, o una empresa mixta o de colaboración público-privada (véase García-Valiñas et al. 2013 para una descripción más detallada del marco legal referente a la externalización). Hay que señalar que únicamente puede ser privatizada la gestión, puesto que la propiedad de los activos debe permanecer, en todo momento, en el dominio público. Además, en ausencia de un marco legal u organismo regulador que establezca ciertos criterios y directrices comunes en relación a las tarifas de agua urbana, los ayuntamientos son, en última instancia, los encargados de la toma de decisiones tanto por lo que se refiere a la fijación de precios, como por lo que respecta a los sistemas de control y monitorización del servicio para asegurar unas adecuadas condiciones de suministro (García-Valiñas et al. 2013).

La elevada descentralización en la toma de decisiones ha derivado en la actualidad en una extraordinaria diversidad de sistemas tarifarios (González-Gómez et al., 2012), siendo los más comunes aquellos que distinguen en la tarifa un término fijo y un componente variable con estructuras de precio crecientes según bloques de

[2] Ratio entre la abstracción anual de agua con respecto a los recursos renovables a largo plazo.

consumo (AEAS-AGA 2013; AEAS, 2013). Cabe asimismo señalar que, a pesar de los esfuerzos realizados por las Administraciones Públicas en los últimos años (European Environmental Agency, 2013), los precios en España continúan sin reflejar el coste total de provisión del servicio, los costes de uso y los costes medioambientales asociados al recurso (AEAS-AGA, 2013; García-Rubio et al., 2015), como sería de esperar de la implementación de la DMA. Por otro lado, el hecho de que las redes de distribución se encuentran notablemente infrafinanciadas, envejecidas y en pobres condiciones de mantenimiento -con niveles de agua no contabilizada que superan el 25% (García-Rubio et al. 2015)- puede suponer una presión adicional para el aumento progresivo de los precios a lo largo del tiempo. Estos rasgos, así como el contexto institucional específico y el marco regulatorio español, que será recogido de forma más exhaustiva a lo largo de la presente tesis doctoral, lo convierten en un marco interesante para el estudio de las cuestiones abordadas en esta disertación.

A continuación se presenta un breve resumen de los cuatro capítulos que componen esta tesis doctoral; se presta especial atención al objetivo y las contribuciones de cada uno de ellos.

Resumen

Capítulo 1: Estimación de la demanda residencial de agua. Consideración de los patrones de sustitución y efectos no lineales en el ingreso.

El primer capítulo de esta tesis doctoral tiene por objeto contribuir a la modelización de la demanda de agua residencial. Tal como se ha mencionado anteriormente en esta introducción, las políticas de demanda y, en especial, de precios, se han revelado esenciales a la hora de lidiar con los retos de futuro derivados de la creciente escasez hídrica. Dado que el análisis de sus efectos pasa necesariamente por una adecuada modelización, la obtención de estimaciones precisas de elasticidades y patrones de demanda ha recibido una amplia atención por parte de los economistas medioambientales.

En la literatura de la demanda de agua residencial, la mayor parte de las estimaciones de elasticidad precio e ingreso se han venido apoyando en la utilización de modelos uniecuacionales, basados implícitamente en los supuestos de separabilidad débil del consumo de agua – i.e., la demanda de agua no depende del precio de los bienes consumidos por el hogar – y en la existencia de efectos lineales con respecto al ingreso – i.e., un incremento en el ingreso ejerce el mismo efecto sobre el consumo de agua, independientemente de la parte de la distribución de ingreso en la que se sitúe el hogar.

Dichos supuestos teóricos introducen ciertas limitaciones en la estimación de la demanda; sin embargo, su validez no se ha verificado con anterioridad. En este contexto, el objetivo de este ensayo es proponer un sistema de estimación de demanda más flexible, el *Quadratic Almost Ideal Demand System* (QUAIDS) (Banks et al. 1997) que, entre otras particularidades, presenta dos ventajas principales frente a las especificaciones funcionales empleadas con anterioridad. En primer lugar, dado que la demanda se estima a través un sistema de ecuaciones, permite tener en cuenta posibles patrones de sustitución y complementariedad entre los bienes consumidos en el hogar, y estimar las elasticidades cruzadas. En segundo lugar, incorpora un término cuadrático en el logaritmo del gasto, lo que permite considerar la existencia de curvatura en las curvas de Engel de los bienes incluidos en el sistema. Asimismo, puesto que este modelo satisface los principios fundamentales de la demanda - i.e., aditividad e integrabilidad-, es de esperar que ofrezca estimaciones más precisas de los efectos de las políticas públicas relacionadas con precios e impuestos.

En el análisis empírico de este trabajo, la información sobre gasto se obtiene de la Encuesta de Presupuestos Familiares (EPS) de España para el periodo 2006-2012, un panel rotatorio de 151.000 hogares (entre 19.000 y 23.000 por año) que constituye una muestra representativa de la población española y contiene información de hasta 489 bienes. Por su parte, la información sobre la evolución de los precios, desglosados a nivel provincial y al mayor nivel de desagregación posible (en 44 subgrupos de bienes), se obtiene de las estadísticas de los Índices de Precios al Consumo (IPC), mientras que el precio medio del agua pagado por el hogar proviene de la muestra de hogares de las EPS referida anteriormente. Con fines de modelización, se supone un proceso de asignación presupuestaria en dos etapas: los hogares primero eligen cómo distribuir su gasto total entre bienes duraderos y no duraderos y, en una segunda etapa, reparten este último entre diversos tipos de bienes no duraderos compuestos – comida y bebida, agua, energía, resto de gastos del hogar y otros bienes. Las estimaciones se obtienen utilizando el Estimador Mínimo Cuadrático Iterado (EMCI) propuesto por Blundell y Robin (1999); la posible endogeneidad de las variables gasto total y precio del agua se aborda mediante técnicas de instrumentación no lineales propuestas por los mismos autores y el enfoque de regresión aumentada de Hausman (1978) y Holly y Sargan (1982).

Los resultados obtenidos apuntan a que el agua es un bien normal, una necesidad (con una elasticidad ingreso a largo plazo de -0.974) cuya demanda es relativamente inelástica al precio (con elasticidad precio a largo plazo 0.108). Además, se desvela la existencia de no linealidades en las curvas de Engel para la demanda residencial de agua, así como relaciones de sustitución y complementariedad con otros grupos de bienes no duraderos, si bien, el agua no parece tener buenos sustitutos.

De este modo, las especificaciones funcionales que se han venido utilizando de forma habitual –i.e. lineal, logarítmica y doble logarítmica-, estarían imponiendo importantes rigideces en la estimación de la demanda de agua. Por tanto, cabe esperar que QUAIDS ofrezca estimaciones más consistentes con los patrones observados en la muestra. Para finalizar se ofrece una comparación de QUAIDS con las especificaciones funcionales más habituales mencionadas anteriormente. La evidencia empírica apunta a que las predicciones obtenidas con QUAIDS se asemejan más a los datos observados (mejor bondad del ajuste) y, además, recogen mejor la asimetría apreciada en la distribución de la demanda de agua, lo que supone una notable mejora frente a otras especificaciones.

Por tanto, de nuestros resultados se concluye que la demanda de agua debería estimarse de forma conjunta con el resto de bienes consumidos en el hogar, utilizando modelos más compatibles con la teoría económica del comportamiento del consumidor. Asimismo, la utilización del modelo propuesto en este artículo junto con la información de las Encuestas de Presupuestos Familiares, disponibles actualmente en un gran número de países, permitiría obtener predicciones nacionales y regionales capaces de informar de forma más precisa las decisiones de política pública adoptadas respecto a la gestión de los recursos hídricos.

Capítulo 2: Una propuesta para el análisis del escalado de precios en las tarifas de agua. El impacto de la Directiva Marco del Agua en España.

Las tarifas en bloques crecientes, en las que el precio por m^3 se incrementa con el nivel de consumo, son utilizadas en la gestión del servicio de agua urbana cada vez con mayor frecuencia. El objetivo de estas estructuras de precios es introducir un cierto nivel de escalado o progresividad en el precio del agua que permita la consecución simultánea de los objetivos de eficiencia, igualdad, sostenibilidad y asequibilidad en la gestión del agua urbana (Olmstead et al. 20017). Sin embargo, a pesar de su extendido uso, hasta el momento no se ha desarrollado una métrica que permita evaluar el nivel de escalado introducido en una determinada estructura de precios. El objetivo de este trabajo es cubrir dicha laguna al proponer una métrica, que permita medir y comparar el grado de progresividad en las tarifas de agua a nivel de la unidad de gestión pertinente, el municipio en este caso. La medida propuesta debe ser capaz de evaluar el nivel de progresividad de la tarifa independientemente de la estructura de precios empleada - es decir, de la presencia de mínimos exentos, número y tamaño de los bloques, y la existencia y magnitud del componente fijo de la tarifa-, y permitir, además, una total comparabilidad entre las jurisdicciones.

En particular, se proponen dos indicadores; uno tendría en cuenta la influencia del componente fijo de la tarifa, mientras que el otro incluiría únicamente la parte variable. Adicionalmente, con el fin de ilustrar la utilidad de la medida propuesta, se realiza un análisis de la evolución de las tarifas de agua en España tras la implementación de la DMA, así como de los factores políticos, medioambientales y empresariales que han marcado dicha evolución. Con este propósito, se utiliza una muestra de 952 municipios en España, para los años 2000 y 2014, es decir, antes y después de la implementación de la DMA.

Tras la entrada en vigor de la DMA, los principales órganos asesores españoles recomendaron a los municipios la implementación de tarifas en forma de bloques crecientes como las anteriormente descritas. Asimismo, se incentivó que se incrementara el nivel de escalado o progresividad en las tarifas. Como consecuencia, sería de esperar que se hubiera producido un aumento en la progresividad de las tarifas aplicadas.

De hecho, los resultados de este capítulo parecen sugerir que los gobiernos locales trataron de incrementar la progresividad de las tarifas a través de aumentos de precios más acusados en los niveles de consumo más elevados. Sin embargo, se encuentra que los principales aumentos tuvieron lugar en aquellos rangos de consumo que presentaban una menor proporción de consumidores, siendo, por tanto, de efecto limitado. No obstante, se observa que, contrariamente a lo que cabría esperar de la implementación de la DMA, se ha producido una caída en el nivel de progresividad introducido en la parte variable de la tarifa. Además, cuando se toma en consideración el efecto del componente fijo, se observa que las tarifas en España son, en media, regresivas en el consumo – el precio por metro cúbico es decreciente-, si bien es cierto que eran menos regresivas en 2014, tras la implantación de la DMA, que en el año 2000. Asimismo, los resultados obtenidos sugieren que algunos factores políticos e ideológicos, las características socioeconómicas del municipio, su actividad turística y la propiedad del gestor del servicio, afectan a la probabilidad de adoptar tarifas más progresivas.

Finalmente, se confía en que las métricas propuestas en este ensayo pongan a disposición de los responsables de políticas públicas las herramientas necesarias para efectuar un sólido análisis que permita evaluación de los esfuerzos llevados a cabo por países y municipios en relación a la mejora de las políticas tarifarias.

Capítulo 3: El papel de las actitudes medioambientales en los comportamientos defensivos que implican externalidades medioambientales negativas: Una aproximación de doble valla aplicada a la demanda de agua embotellada.

La motivación del tercer capítulo surge del hecho de que ciertas políticas de gestión urbana de agua podrían estar afectando, en última instancia, a otras decisiones tomadas en el hogar. Por ejemplo, una insuficiente calidad del agua del grifo, así como otros factores relacionados con la gestión de los servicios urbanos de agua, podrían resultar en la adopción de ciertos comportamientos defensivos, tales como la instalación de sistemas de filtrado o el consumo habitual de agua embotellada.

El consumo de agua embotellada es un comportamiento defensivo que implica significativas externalidades medioambientales, y que cuenta con sustitutos más respetuosos con el medio ambiente (e.g. dispositivos de filtrado). En este contexto, el presente ensayo tiene un triple objetivo. En primer lugar, se investiga el papel que juegan las actitudes y comportamientos medioambientales de los sujetos, en la decisión de llevar a cabo comportamientos defensivos que conlleven este tipo de externalidades. Para ello, además, consideramos las dos categorías de comportamientos pro-medioambientales diferenciados en la literatura de acuerdo con el nivel de esfuerzo que exigen por parte del individuo: comportamientos puntuales (*efficiency behaviors*)– i.e., instalar ciertos dispositivos ahorradores de agua- y comportamientos continuados (*curtailment behaviors*) – i.e., hábitos diarios que implican sacrificios con el objeto de ahorrar recursos. En segundo lugar, dado que el agua de botella y la del grifo podrían ser bienes complementarios o sustitutivos, se explora el posible impacto sobre la demanda de agua embotellada de ciertas políticas tarifarias y no tarifarias relacionadas con la gestión residencial del agua que actualmente están siendo aplicadas en numerosos centros urbanos en el mundo. En tercer y último lugar, la intención de este ensayo es proponer una estrategia empírica para modelizar adecuadamente aquellos comportamientos defensivos que presenten problemas derivados de una elevada proporción de consumos nulos (respuestas cero).

Con este propósito, se utilizan datos, para el año 2014, de una muestra representativa de 592 hogares de las ciudades de Baza y Guadix, localizadas en la provincia española de Granada. España es el noveno país del mundo en términos de consumo per cápita de agua embotellada y el cuarto entre los países europeos. Además, dado que el área en la que se encuentran Baza y Guadix sufre actualmente de estrés hídrico severo (European Environmental Agency, 2012), es habitual que se lleven a cabo múltiples políticas de conservación del recurso en la región, convirtiéndolo en un interesante contexto para este estudio. Con el objetivo de lidiar con el elevado porcentaje de registros nulos, se propone un enfoque metodológico de doble valla. Esto posibilita modelizar el comportamiento defensivo sin partir de ninguna hipótesis particular sobre las razones por las que los hogares no llevan a cabo dichos comportamientos – i.e, no-participación vs solución de esquina-. Asimismo, este enfoque permite examinar a través de *tests* estadísticos los supuestos distribucionales que subyacen a las mismas, con el objeto de elegir entre especificaciones.

Los resultados obtenidos sugieren que ciertas políticas de demanda, tarifarias y no-tarifarias, tales como incrementos en el precio del agua corriente e interrupciones en el suministro, podrían derivar en un incremento en la demanda del agua embotellada y generarían, por tanto, consecuencias no deseadas sobre el medio ambiente. También se concluye que promover ciertos hábitos pro-medioambientales puede ser muy eficaz para evitar comportamientos defensivos que provoquen externalidades medioambientales negativas. Por último, el análisis sugiere que no abordar de forma adecuada los problemas derivados de la elevada ocurrencia de registros de consumo nulos en la modelización de comportamientos defensivos puede llevar a conclusiones erróneas.

Capítulo 4. Revisandola relación entre titularidad del gestor y el desempeño en la prestación del suministro de agua: ¿Realmente supera la gestión privada a la pública?.

El último ensayo de esta tesis doctoral tiene por objeto revisar la relación entre la titularidad del gestor y el desempeño en la prestación del servicio. Desde los años 70 del siglo pasado, se viene produciendo un intenso debate acerca de qué forma de titularidad del servicio (pública o privada) se muestra superior en términos de eficiencia. Sin embargo, la literatura no ha sido hasta el momento capaz de mostrarse conclusiva al respecto. En este trabajo se pretende arrojar algo de luz a este debate a través de la utilización de una nueva estrategia metodológica basada en un Análisis Envolvente de Datos (DEA), que combina dos aproximaciones empíricas: las funciones de distancia direccionales y las metafronteras.

La contribución del trabajo es doble. En primer lugar, el uso de metafronteras permite discernir si las diferencias de rendimiento encontradas entre la gestión pública y la privada se deben a diferentes capacidades de sus directivos –eficiencia de gestión- o a la existencia de distintas tecnologías productivas afrontadas por las unidades públicas y privadas – eficiencia de propiedad. Esta distinción es importante en la medida en que existen razones para creer que la gestión pública y privada difieren respecto a las restricciones tecnológicas a las que se enfrentan – i.e., marco institucional y legal. En segundo lugar, el uso de funciones de distancia direccionales tiene la ventaja de permitir la evaluación del desempeño a nivel de los distintos factores productivos involucrados en el proceso de producción. Dado que las restricciones tecnológicas pueden afectar de forma diferencial a los distintos factores de producción, la combinación de estos dos enfoques muestra un mayor potencial a la hora de ofrecer nuevas perspectivas frente a estudios previos que han utilizado separadamente metafronteras o funciones distancia direccionales.

El análisis empírico en este trabajo se basa en una muestra de empresas de agua que prestan el servicio en 70 municipios españoles de menos de 50.000 habitantes en el año 2013. En 37 de estos municipios, la provisión del servicio urbano de agua la realiza el propio ayuntamiento o bien una empresa pública, mientras que en el resto de municipios este servicio ha sido externalizado a una empresa privada o de colaboración público-privada. El desempeño en la gestión se evalúa en términos de eficiencia técnica. El proceso productivo se caracteriza por la obtención de dos productos finales (*outputs*) – agua suministrada y población servida – a partir del uso de tres factores productivos (*inputs*); uno fijo – la longitud de la red de distribución - y dos variables – trabajo y costes operativos. El análisis se orienta al input, es decir, analiza la capacidad de la empresa de reducir sus factores productivos manteniendo el mismo nivel de producción.

Los resultados obtenidos son los siguientes. Cuando se considera el escenario convencional, utilizado en la mayoría de estudios previos, en que la eficiencia se evalúa en una dirección en la que se reducen de forma proporcional todos los *inputs* - eficiencia radial-, no se observan diferencias estadísticamente significativas en el desempeño de operadores públicos y privados. Sin embargo, al utilizar la aproximación metodológica propuesta en este trabajo, se obtiene una perspectiva sustancialmente distinta. Por una parte, los resultados sugieren que la tecnología de los operadores privados es más eficiente en la gestión del factor trabajo, mientras que la tecnología de las unidades públicas lo es en la gestión de los costes operativos. Estos resultados se explican por diversos factores. En primer lugar, por el hecho de que los gestores públicos afrontan habitualmente marcos laborales más restrictivos y mayores niveles de absentismo (Meier and O'Toole 2011), así como por otros factores políticos e institucionales específicamente relacionados con este tipo de gestión. Por su parte, en lo que respecta a los costes operativos, las unidades de gestión pública tienden a compartir dichos costes con el resto de servicios provistos por la misma administración pública, especialmente cuando el servicio lo gestiona el propio ayuntamiento. Además, los operadores privados suelen afrontar mayores dificultades en la gestión de los costes operativos como consecuencia de que habitualmente operan en ambientes de gestión más complejos (González-Gómez et al., 2011).

En síntesis, los resultados obtenidos apuntan a que la estrategia empírica propuesta resulta satisfactoria a la hora de desvelar ciertos aspectos que permanecían ocultos a las medidas de desempeño utilizadas convencionalmente. Esto tiene importantes implicaciones para el diseño de políticas públicas, ya que propone una herramienta de gestión útil para un elevado número de grupos de interés – desde directivos hasta organismos reguladores -, capacitándolos para tomar decisiones más informadas acerca de qué prácticas pueden resultar favorables para la industria. Asimismo, este trabajo contribuye al debate, anteriormente mencionado, sobre la relación entre propiedad del gestor y eficiencia de los servicios urbanos de agua.

References

AEAS (2014). Guía de Tarifas de los Servicios de Abastecimiento y Saneamiento de Agua. Madrid, Spain.

AEAS-AGA (2012). XII Edición "Encuesta de Suministro de Agua Potable y Saneamiento en España ".Can be accessed in: http://www.asoaeas.com/sites/default/files/Documentos/Nota.

Arnell, N. (2000). Impact of climate change on global water resources: Volume 2, unmitigated emis-sions. Report to Department of the Environment, Transport and the Regions. University of Southampton, Southampton, United Kingdom, 53 pp.

Arnell, N. (1999). Climate change and global water resources. Global Environmental Change, 9:S31–S49.

Banks, J., Blundell, R., and Lewbel, A. (1997). Quadratic engel curves and consumer demand. Review of Economics and Statistics, 79:527–539.

Biswas, A. K. and Tortajada, C. (2009). Water Management in 2020 and Beyond, chapter Changing Global Water Management Landscape 1–33. Springer.

Blundell, R. and Robin, J.-M. (1999). Estimation in large and disaggregated demand systems: An estimator for conditionally linear systems. Journal of Applied Econometrics, 14:209–232.

European Environmental Agency (2013). Assessment of cost recovery through water pricing. EEA Technical Report No 16/2013. Copenhagen, Denmark.

European Environmental Agency (2012). Towards efficient use of water resources in Europe. EEA Report No 1/2012. Copenhagen, Denmark.

García-Rubio, M.A., Ruiz-Villaverde, A., González-Gómez, F. (2015). Urban water tariffs in Spain: what needs to be done?. Water 7 (4): 1456-1479.

García-Valíñas, M.A., González-Gómez, F., and Picazo-Tadeo, A.J. (2013). Is the price of water of residential use related to provider ownership? Empirical evidence from Spain. Utilities Policy 24(1): 59-69.

González-Gómez, F., García-Rubio, M.A. and Guardiola, J. (2012). Urban water service policies and management in Spain: Pending Issues. International Journal of Water Resources Development 28(1): 89–106.

Hausman, J. A. (1978). Specification tests in econometrics. Econometrica 46:1251–1271.

Holly, A. and Sargan, D. (1982). Testing for exogeneity in a limited information framework. Cahiers de Recherches 8204, University of Lausanne.

IPCC (2014). Climate change 2014: Synthesis report. contribution of working groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [core writing team, R.K. Pachauri and L.A. Meyer (eds.)]. Technical report, IPCC, Geneva, Switzerland.

Jacob, D.; Petersen, J.; Eggert, B.; Alias, A.; Christensen, O.B.; Bouwer, L.M.; Braun, A.; Colette, A.; Déqué, M.; Georgievski, G.; et al. (2014) EURO-CORDEX: New high-resolution climate change projections for European impact research. Regional Environmental Change 14: 563–578.

Meier, K.J., and O'Toole, L.J. (2011). Comparing public and private management: Theoretical expectations. Journal of Public Administration Research and Theory 21: 283–299.

OECD (2011). Managing Water for All. An OECD Perspective on pricing and financing. OECD Publishing, Paris.

OECD (2003). Social issues in the provision and pricing of water services. OECD Publishing, Paris.

Olmstead, S. M. and Stavins, R. N. (2009). Comparing price and nonprice approaches to urban water conservation. Water Resources Research, 45(4).

Olmstead, S. M., Hanemann, W. M., and Stavins, R. N. (2007). Water demand under alternative price structures. Journal of Environmental Economics and Management, 54:181–198.

Schlosser, C., Strzepek, K., Gao, X., Gueneau, A., Fant, C., Paltsev, S., Rasheed, B., Smith-Greico, T., B., E., Jacoby, H., and Reilly, J. (2014). The future of global water stress: An integrated assessment. Technical report, Joint programme report series, MIT Joint Programme on the Science and Policy of Global Change.

Suárez-Varela, M., Martínez-Espiñeira, R., and González-Gómez, F. (2015). An analysis of the price escalation of non-linear water tariffs for domestic uses in Spain. Utilities Policy 34(2015): 82-93.

UN Population Division (2015). World population prospects 2015. Technical report, United Nations, Department of Economic and Social Affairs, Population Division.

UN Population Division (2014). World urbanization prospects: The 2014 revision, highlights (st/esa/ser.a/352). Technical report, United Nations, Department of Economic and Social Affairs, Population Division.

UNESCO (2016). Water, Megacities and Global Change. United Nations Educational, Scientific and Cultural Organization. 7 Place de Fontenoy, 75352 Paris 07 SP, France.

World Economic Forum (2017). The global risks report 2017. 12th edition. World Economic Forum. Cologny/Geneva. Switzerland.

Essay 1: Estimation of residental water demand. Accounting for susbstitution patterns and non-linear income effects.

Abstract

Most estimations of income and price elasticities for residential water are based on single-equation models that rely on assumptions of the separability of water from other goods and linear income effects on water consumption, whether linearities in Engel curves or income elasticities that are constant for every level of income. In this paper, we relax these assumptions by using a more flexible system of demand estimation, the Quadratic Almost Ideal Demand System (QUAIDS) (Banks et al., 1997) and reveal the existence in our sample of substitution and complementary patterns as well as non-linearities in Engel curves for water consumption. Moreover, in this context the QUAIDS functional specification is expected to be more consistent with observed consumer behavior. Our results seem to confirm this expectation; when compared to the linear, log-linear and double-log models commonly used

in water demand estimation, QUAIDS seems to produce a better overall fit and a better fit to the asymmetric shape of the real distribution of water consumption. Therefore, in order to avoid bias in the estimates, and when the necessary information is available, water demand should be jointly estimated with the rest of the goods consumed in the household and higher order income terms should be considered.

1.1. Introduction

Water scarcity is expected to pose a major challenge in the near future. In fact, it is predicted that by 2050 more than half of the global population will live in regions suffering from at least moderate water stress (Arnell, 1999, 2000; Schlosser et al., 2014). Furthermore, by that time, the population is predicted to have risen from the current 7.3 billion people to 9.7 billion (UN Population Division, 2015). This unprecedented exponential population growth, along with rising incomes and economic development, is forecast to increase global water needs for residential uses and food production. Likewise, demand for energy generation and industry is also projected to undergo substantial growth. Increases of between 50 and 70% are estimated for industrial uses and up to 85% for energy production (World Bank Group, 2016). Moreover, with a predicted 66% share of the world's population residing in urban areas by 2050 (UN Population Division, 2014), urbanization trends represent an additional pressure on urban water management (Biswas and Tortajada, 2009).

Water supply is expected to be affected as well. As a consequence of rising temperature levels and of increasingly extreme and variable rainfall, both droughts and floods are projected to become more common (IPCC, 2014). Furthermore, rising sea levels could result in the salinization of aquifers, leading to a decrease in available freshwater. Consequently, a supply diminished by climate change, becoming more volatile and increasingly difficult to predict, will further exacerbate water availability problems. In economic and social terms, the costs of water scarcity could reach 6% of GDP in some regions, as well as triggering large-scale migration flows and civil conflicts (World Bank Group, 2016; World Economic Forum, 2016). In this context, efficient and sustainable management of water resources emerges as a paramount issue.

Traditionally, water shortage problems have been addressed by increasing supply. The most common solutions include building infrastructure such as dams or canals, recycling wastewater or, more recently, desalination. On their own, however, supply interventions are not enough to solve the problem. Moreover, when such measures are implemented without adequate economic incentives for water conservation, they have often proved to induce new demand (World Bank Group, 2016), resulting in faster resource depletion. With little margin on the supply side, demand-side policies are thus revealed as the most effective means to address water scarcity. Of these[3], pricing has been found to be the most cost-effective tool for promoting water conservation (Olmstead and Stavins, 2009).

In this sense, accurate estimates of demand patterns prove crucial to undertake precise diagnoses about the effectiveness and consequences of certain commonly used water pricing policies. Accordingly, the literature on estimating residential water demand is vast, and numerous theoretical specifications have been used, with the linear, log-linear and double log being the most common. However, despite the considerable attention that this subject has attracted, there has been little debate on which specification is best in terms of fit and performance (European Commission, 2015).

Moreover, most estimations of income and price elasticities for residential water are based on single-equation models that rely on assumptions of weak separability of water consumption from other goods—i.e. assuming that household water consumption does not depend on the price of other goods— (European Commission, 2015) and the existence of linear income effects on water consumption. The usual underlying argument for imposing separability is that it is feasible given that there are no good substitutes for indoor water and both household habits and the stock of durables related to water consumption are unlikely to change in the short term (Arbués et al., 2003). However, those arguments have never been tested against empirical evidence, probably because there have been relatively few databases to date that contain information on water consumption and consumption of other goods at the household level. Furthermore, the abovementioned arguments do not take into account several features of residential water demand. Besides indoor consumption, domestic water also includes some outdoor uses. In addition, although normally treated as a homogenous good, water is in fact a composite commodity that satisfies different needs. In this vein, while water for drinking, cooking or personal hygiene can be considered a basic need, water for filling swimming pools or watering gardens could be regarded as a type of luxury good. Other uses, such as those related to laundry and car washing would fall into an intermediate

[3] Demand-side policies include both pricing policies and non-pricing policies (such as rationing, information campaigns that promote conservation or the installation of new water-saving technologies).

category. Thus, whereas there may be no substitutes for some water uses (mainly the basic and some intermediate uses), consumption surpassing a certain threshold is more easily substitutable. Moreover, residential water may be a substitute for other commodities and activities. In addition, water could be complementary to the consumption of other goods, even at the basic consumption levels. In summary, beyond the basic threshold required to satisfy basic needs, water demand may reflect a reaction to changes in the prices of other goods and, conversely, consumption for other goods may vary in accordance with the price of water. With respect to the treatment of income, as is explained more thoroughly in next section, patterns displayed in the most typically used models are also rather simple and most often not compatible with basic assumptions of consumer theory (e.g. the adding-up restriction).

In this paper, we use a less restrictive system of demand estimation, the Quadratic Almost Ideal Demand System (QUAIDS) (Banks et al., 1997). As opposed to most common models, QUAIDS displays several advantages. First, QUAIDS does not impose separability, thus incorporating all the information on income and substitution effects (stemming from variations in relative prices) contained in cross-price elasticities. Moreover, it accounts for the curvature of Engel curves[4] by including quadratic terms in the logarithm of expenditure. This has important implications in the sense that income effects are now allowed to vary across the income distribution. Therefore, this paper's use of QUAIDS allows us to make several contributions. On the one hand, we can test empirically whether, as usually assumed, water consumption is independent of relative changes in the prices of other goods consumed within the household. On the other hand, we can also test for non-linear income effects. Moreover, it displays two main advantages that can contribute to enhance the effectiveness of water demand policies. First, it satisfies the fundamental principles of consumer theory, so it is expected offer improved estimates of demand. Moreover, since it satisfies integrability[5], it allows to ascertain accurately the welfare effects of policies affecting prices and taxes in terms of issues such as economic efficiency, effectiveness (whether it would actually reduce consumption) and social and equity concerns (i.e. water poverty).

For this purpose, we use data from the Spanish Consumer Expenditure Survey (EPF or "Encuesta de Presupuestos Familiares" in Spanish), a rotating panel of 151,068 observations (between 19,000 and 23,000 households per year)

[4] In microeconomics, the Engel curve (Engel, 1895) describes the relationship between commodity expenditure and income.

[5] The capacity of retrieving the cost function from the estimated demand system.

representative of the Spanish population for the period 2006-2012. Given that this dataset includes household expenditures for up to 489 goods, the interaction of water demand with all other goods can be explored.

In Spain, the average annual per capita water consumption in the period under study is 43.2m^3, representing an average of 118 liters per person per day[6]. According to WHO/UNICEF (2008), the basic or "lifeline" level of domestic water use, covering drinking and strictly subsistence level sanitation and personal hygiene, stands at between 15 and 25 liters per person per day. Other authors such as Gleick (1996) recommend a minimum amount of 50 liters to adequately cover requirements relating to drinking, hygiene, sanitation and food preparation. Compared to that basic level, Spanish average consumption is more than double the recommended amount, suggesting that it may be satisfying other higher-level uses more susceptible to relative changes in price.

Our results suggest that, as indicated by some authors (Arbués et al., 2003), water has no good substitutes among non-durable commodities; however, it does seem to be a substitute for and a complement to other goods consumed within the household. Moreover, evidence suggests that Engel curves for water are not linear. In this context, it would seem that previously used models and specifications impose implausible restrictions on water demand. Conversely, QUAIDS is expected to be more consistent with consumer theory, thus offering a richer picture of consumers' expenditure behavior. As a final step, we estimate the linear, log-linear and double-log models with our data and compare the results to those of the QUAIDS specification. The fact that, as shown at the end of the paper, QUAIDS displays better overall fit and a significantly better fit to the shape and asymmetry of the real distribution for water consumption, seems to confirm our supposition that this system can better approximate the consumption patterns observed in the data. Thus, our results appear to indicate that, in order to avoid bias in the estimates, and when the necessary information is available, water demand should be estimated in relation to the demand for the other goods consumed in the household, and non-linear behavior with respect to income should also be accounted for. With the increasing availability of country-representative household budget surveys in many countries of the world, as well as detailed microdatasets on household expenditures, the model proposed in this paper could be used to further exploit those datasets, as

[6] This amount is obtained from our own database, which is a representative sample of the Spanish population. However, the figure is not very different from the consumption estimated by other organizations that produce statistics on water consumption in Spain. The most widely cited, the annual Spanish Association of Water Supply and Sanitation survey, reported an average of 126 liters per person per day in 2012 (AEAS-AGA, 2012).

well as to obtain country and regional prospects necessary to inform long-run policy decisions aimed at addressing water scarcity.

The remainder of the paper is organized as follows. Section 1.2. reviews the literature on residential water demand, while section 1.3. outlines the model and data used in the paper. Finally, section 1.4. presents and discusses the main results and conclusions are drawn in section 1.5.

1.2. Modeling residential water demand

Analysis of water demand has long been a focus of study. An accurate measurement of water consumption patterns is essential in order to assess the effects of certain public policies aimed at controlling demand. Consequently, there is an extensive body of literature on this subject, which discusses a significant number of issues and approaches. In fact, numerous attempts have been made to synthesize the information, including some reviews (see Arbués et al., 2003; Nauges and Whittington, 2010 and Worthington and Hoffman, 2008) and a few meta-analyses (Dalhuisen et al., 2003; Sebri, 2016; Espey et al., 1997). Some of the issues addressed in the literature that we cover here range from demand and price specification to simultaneous determination of prices and quantities, particularly in the case of block tariffs.

Concerning functional specifications, although some more complex functions such as Stone-Geary (Al-Quanibet and Johnston, 1985; Gaudin et al., 2001; Martínez-Espiñeira and Nauges, 2004; Madhoo, 2009; Dharmaratna and Harris, 2010) have been applied, the bulk of studies on water demand estimation rely on single-equation models that assume linear, log-linear or double logarithmic forms (Arbués et al., 2003; European Commission, 2015). The drawback of these functional specifications is that they impose implausible restrictions on water demand. On the one hand, linear models are often criticized on the grounds that marginal effects (that is, the change in quantity demanded as a consequence of a unit change in price) are assumed to be constant, indirectly yielding price elasticities that are smaller for lower price levels (Arbués et al., 2003). Moreover, the fact that water is an essential good needed for survival seems to be at odds with a negative linear relationship between quantity and price (Al-Quanibet and Johnston, 1985) [7].

Another common specification, the double log model, also presents some shortcomings. Although the simplicity of interpreting the estimated coefficients directly as elasticities have made it the most widely-used model in water demand estimation (European Commission, 2015), price elasticities are assumed to be constant at every price level. Moreover, the fact that it violates the adding-up restriction[8] deriving from the existence of a budget constraint makes it inconsistent with consumer theory, leading to important distortions in the prediction of demand patterns (Deaton and Muellbauer, 1980b). Likewise, regardless of the functional form, the use of a single equation in most studies relies on the non-empirically tested assumption that water demand is weakly separable. This implies ignoring the existence of income and substitution effects deriving from changes in relative prices (Deaton and Muellbauer, 1980b), which also generates bias in consumption pattern estimates. On the other hand, the few existing studies that use the more complex Stone-Geary function (Al-Quanibet and Johnston, 1985; Gaudin et al., 2001; Martínez-Espiñeira and Nauges, 2004; Madhoo, 2009; Dharmaratna and Harris, 2010) offer the advantage that elasticities are allowed to vary across price (non-constant price elasticities) and to some extent across quantities[9]. However, this function still imposes some severe restrictions such as strong separability and positive budget elasticity[10] (Deaton and Muellbauer, 1980b; Stone, 1954).

With respect to income, the typical linear and double log models display rather simple income patterns, imposing constant marginal effects of income on water consumption or constant income elasticities, respectively, and, as mentioned above, violating the adding-up restriction. Even the more sophisticated functional forms such as the Almost Ideal Demand System (Deaton and Muellbauer, 1980a), which has proved more consistent with utility theory (see Deaton and Muellbauer, 1980a and Deaton and Muellbauer, 1980b and for more details), still assume that the

[7] Note that under this shape, demand will eventually intersect the price axis leading to a point at which there is no demand at all for water.

[8] Under the adding-up condition, consumers have an exogenous budget to be allocated among the different goods that they purchase, such that the total sum of the amounts spent on each of the goods does not exceed the total budget at their disposal.

[9] By modeling a minimum amount that is assumed to be insensitive to prices (Stone, 1954; Martínez-Espiñeira and Nauges, 2004).

[10] This last assumption should not be very problematic, as residential water demand is usually found to be a normal good (Arbués et al., 2003; Worthington and Hoffman, 2008).

shares of total expenditure devoted to different goods (also known as budget shares) are linear in the log of total expenditure. As shown by empirical studies (see for example, Atkinson et al., 1990; Blundell et al., 1993; Hausman et al., 1995; Hildenbrand, 1994; Lewbel, 1991), those income patterns do not fully capture consumer behavior as they do not account for the existence of non-linearities in the Engel curves for some goods. Higher income terms are thus required to offer an adequate picture of consumption patterns (Banks et al., 1997).

As opposed to these models, the Quadratic Almost Ideal Demand System (QUAIDS) we use in this paper offers several advantages. First, it does not impose separability, thus allowing for the possible impact of changes in relative prices. Moreover, a quadratic term in the logarithm of expenditure is included to test for the likely curvature of Engel curves. This implies that income is now allowed to vary across income groups, making it more consistent with empirically observed consumption patterns (Banks et al., 1997) [11]. In addition, QUAIDS uses total expenditure instead of income itself. For a number of practical as well as theoretical reasons, economists generally prefer information on total expenditure (when available) rather than income (see Deaton and Zaidi, 2002 p.11-13 for further clarification on this issue). Finally, although not directly relevant to this study, QUAIDS satisfies integrability conditions, which enables the analysis of the welfare effects of certain public policies related to price and taxes. As a consequence of all these properties, QUAIDS exhibits much sounder theoretical foundations and provides a more realistic picture of consumer behavior as compared to previously used models.

Price specification and endogeneity are also among the main concerns in the residential water demand literature. There is a general trend in water management to introduce non-linear tariffs in the form of Increasing Block Rates (IBRs henceforth) (Olmstead et al., 2007), which involve marginal price increases in blocks corresponding to different ranges of consumption. Particularly in Spain, where the data for our empirical implementation comes from, the use of IBRs has become increasingly widespread in the last decade (AEAS-AGA, 2013). The modeling of water demand specifications under this type of tariff has generated a great deal of debate in the literature. In the presence of perfect information and facing the non-linear budget constraint arising from a block pricing structure, standard economic theory predicts that consumers will optimize their consumption in accordance with

[11] Contrary to the abovementioned models that assume linearities in Engel curves or income elasticities that are constant for every level of income.

the actual marginal price structure of the tariff they have to pay[12]. Under this assumption, common practice in early studies involved the use of marginal price as the price variable to which the consumer should be responding. Taylor (1975) was the first to suggest that a sole marginal price may not be enough to capture demand decisions made by consumers, proposing instead that both average and marginal prices should be accounted for simultaneously (Taylor, 1975). A year later, building on Taylor's work, Nordin (1976) proposed that if the changes in consumer surplus as a result of block pricing were to be accounted for, it was more appropriate to include marginal price along with a difference variable to control for the income effect of consumers moving from one block to another.

However, although these specifications have been widely used, empirical evidence shows that most information on tariff structure is far from perfect (Nieswiadomy and Molina, 1989; Nataraj and Hanneman, 2011; Pérez-Urdiales et al., 2015), and even if it were, consumers find it difficult to understand complex pricing and non-linear structures (Nieswiadomy and Molina, 1989; De Bartolome, 1995). The existence of unexpected shocks to both income (Saez 1999, 2010) and demand (Borenstein, 2009) that are beyond the consumer's control, reinforces the difficulty in assuming perfect optimizing behavior. In this context, it seems more logical to assume that consumers will respond to the price they perceive. Indeed, some theoretical models have attempted to accommodate deviation from standard economic theory. For instance, Saez (1999, 2010) and (Borenstein, 2009) predicted that when random shocks to income and demand, respectively, are allowed, consumers are predicted to respond not to the actual marginal price, but rather to the expected marginal price (Saez, 1999; Borenstein, 2009) or even less precise information about marginal price (Borenstein, 2009). Alternatively, Liebman and Zeckhauser (2004) relax the assumption of consumers' perfect understanding of the nature of multiple block tariffs, finding that if consumers cannot be assumed to be perfect optimizers, the perceived price corresponds to the average price at their consumption level.

Ultimately, the shape of the perceived price will diverge according to the assumptions made about consumer behavior. In recent years, discussion has mostly focused on whether to use the average or marginal price as the price variable. Empirical literature is also inconclusive as to which price specification should be

[12] Consumer behavior under structures of this type is of interest not only in the water demand literature; non-linear schedules are also very common in taxes and other markets such as electricity, natural gas and mobile phones. Therefore, some of the studies cited here may be related to these fields.

used under non-linear structures. While numerous empirical (Shin, 1985; Liebman and Zeckhauser, 2004), pseudo-experimental (Ito, 2014; Wichman, 2014) and experimental studies (De Bartolome, 1995) provide evidence that consumers respond to the average price, others support marginal price as the consumer response variable (Nataraj and Hanneman, 2011; Baerenklau et al., 2014).

Finally, non-linear pricing is also at the core of another key empirical debate in water demand estimation. In block pricing, simultaneity bias may arise from the fact that quantities are affected by prices, which in turn are dependent on the quantity consumed. The most common ways of addressing this issue involve using either the Discrete Continuous Choice (DCC) Model proposed by Hewitt and Hanemann (1995) or econometric techniques such as instrumental variables (IV), two-step least squares (2SLS) or three-step least squares (3SLS), and the Generalized Method of Moments (GMM) (Olmstead, 2009). More recently, some authors have also used quasi-experimental designs as a means to address endogeneity.

Estimates of price elasticity have almost unanimously proved residential water consumption to be price inelastic, with the most usual range being from 0 to 0.5 in the case of short-run elasticities and from -0.5 to -1 for their long-run counterparts (Worthington and Hoffman, 2008). This is because, in the long run, durable water-saving equipment can be adjusted following a change in water prices, making demand more price responsive. Moreover, the price elasticities obtained are supposedly independent of the functional specification (i.e. linear, log-linear, etc.) or the estimation technique (Worthington and Hoffman, 2008), although some models such as the DCC have been found to yield higher elasticities (Sebri, 2016; Dalhuisen et al., 2003). With respect to income, reported elasticities are usually positive and lower (in absolute terms) than price elasticities (Worthington and Hoffman, 2008).

1.3. Model and data

1.3.1. Model

The model we estimate is a Quadratic Almost Ideal Demand System (see Banks et al. (1997) for a comprehensive description), an extension of the Almost Ideal

Demand System (Deaton and Muellbauer, 1980b) , which incorporates a quadratic term in the log of total expenditure to account for non-linearities in Engel curves.

As is standard practice in demand systems estimation (see Blundell, 1988), we assume consumers follow a two-step budgeting procedure in which households first decide the budget they allocate to leisure, savings and durable and non-durable goods, and then total expenditure on non-durables is divided among several groups of non-durable commodities[13].

To derive the QUAIDS for the N non-durable goods consumed by the household, we first define the indirect utility function of a household *h* as follows:

$$V^h(p,m) = \left[\left(\frac{ln\ m - \ln a(p)}{b(p)}\right)^{-1} - \lambda(p)\right]^{-1} \tag{1}$$

where m is total expenditure in goods *i, j= 1, …, N; ln a(p)* and *b(p)* are price indexes from the AIDS model of Deaton and Muellbauer (1980a).

$$\ln a(p) = \alpha_0 + \sum_{i=1}^{N} \alpha_i \ln p_i + \frac{1}{2} \sum_{i=1}^{N}\sum_{j=1}^{N} \ln p_i \ \ln p_j \tag{2}$$

$$b(p) = \prod_{i=1}^{N} {p_i}^{\beta_i} \tag{3}$$

[13] Durable commodities are excluded from the analysis for several reasons. Although expenditures on durable goods are recorded in a given period, the goods themselves are in fact consumed over a longer period of time, which has important implications. As consumption does not coincide with recorded expenses, it cannot be proxied by current expenditure. Moreover, by their nature, durable commodities exhibit a substantial proportion of zero expenditures, which has severe implications for estimation. Concerning water demand estimation, water might be expected to have some substitutes and complements among durable commodities. For instance, faced with an increase in the price of residential water, households may decide to adapt their durable equipment in the long-run by adopting some water-saving technologies such as low-flow taps or dual-flush toilets. Also, water demand can depend on whether the household has certain complementary goods such as washing machines. Unfortunately, our database does not contain information on expenditures on durable commodities related to water consumption. Therefore, including non-durable commodities in the analysis would give rise to the abovementioned complications without significantly contributing to the conclusions of this study.

and λ (p) is an homogeneous function of degree zero in prices p.

$$\lambda(p) = \sum_{i=1}^{N} \lambda_i \ \ln p_i \tag{4}$$

$$\sum_{i=1}^{N} \lambda_i = 0 \tag{5}$$

Applying Roy's identity, budget shares can be written as:

$$w_i = \alpha_i + \sum_{i=1}^{N} \gamma_{ij} \ln p_i + \beta_i \left[\ln \frac{m}{a(p)}\right] + \frac{\lambda_i}{b(p)} \left[\ln \frac{m}{a(p)}\right]^2 \tag{6}$$

where m is total expenditure on all goods in the demand system and w_i is the budget share for good i . *lna(p)*, *b(p)* and *λ (p)* have already been described above.

Economic theory also imposes some restrictions on the parameters of the model. Adding-up requires $\sum \alpha_i = 1$, $\sum \beta_i = 0$, $\sum \gamma_{ij} = 0$ and $\sum \lambda_i = 0$. Homogeneity is satisfied when $\sum \gamma_{ij} = 0$ and symmetry is given by $\gamma_{ij} = \gamma_{ji}$ Moreover, the diagonal elements of the Slutsky or substitution matrix must be nonpositive (Negativity).

The price and budget elasticities of interest to this study are obtained by differentiating equation (6) with respect to *ln m* and *ln pj* respectively.

Budget elasticities are computed as:

$$\epsilon_i = \frac{\mu_i}{w_i} + 1 \tag{7}$$

where

$$\mu_i \equiv \frac{\delta w_i}{\delta \ln m} = \beta_i + \frac{2\lambda_i}{b(p)} \left[\ln \frac{m}{a(p)}\right] \tag{8}$$

The uncompensated own and cross-price elasticities are obtained as follows:

$$\epsilon_{ij} = \frac{\mu_{ij}}{w_i} - \delta_{ij} \tag{9}$$

where δ_{ij} is the Kroneker delta (equal to 1 when *i=j* and *0* otherwise) and

$$\mu_{ij} \equiv \frac{\delta w_i}{\delta \ln p_j} = \gamma_{ij} - \mu_i \left(\alpha_j + \sum_{k=1}^{N} \gamma_{jk} \ln p_k \right) - \frac{\lambda_i \beta_j}{b(p)} \left[\ln \frac{m}{a(p)}\right]^2 \tag{10}$$

Finally, demographic variables are introduced into the model using the translating method (Pollak and Wales, 1978, 1981). With this technique, the intercept is allowed to depend on certain household characteristics that are expected to affect household preferences, by expressing ai parameters as linear functions of those sociodemographic variables.

1.3.2. Data

The data we use in this paper comes from the Spanish Consumer Expenditure Survey (EPF or “Encuesta de Presupuestos Familiares” in Spanish), an annual rotating panel comprising a representative sample of the Spanish population. This survey presents a comprehensive record of household expenditures on up to 489 different goods reported on an annual basis. Furthermore, it contains information on a number of other socioeconomic and household characteristics, as well as some information regarding the location[14] of the household and quantities purchased in physical units for some of the goods (e.g. water). More detailed information about the variables included and sampling procedure can be found in INE (2012).

The data are pooled for the period 2006-2012, providing a sample of 151,068 observations (between 19,000 and 23,000 households per year). Moreover, the fact that, unlike in other countries, the Consumer Expenditure Survey is conducted every year in Spain, prevents problems relating to insufficient micro data and price variation from hindering the identification of demand patterns and elasticities (Pashardes et al., 2014; Nichele and Robin, 1995). As is standard practice in household demand system estimation (see for instance Tiezzi (2005) or Labandeira et al. (2006) for the case of energy demand estimation[15]), non-durable goods are aggregated in broad categories: 1) food, beverage and tobacco (for simplicity, hereafter referred to as 'food'), 2) water supply[16] 3) energy[17]; 4) household expenses other than water and energy[18] and 5) all other commodities. In our model, both energy and other household expenses are separated from food and all other expenses because they are believed to be more closely linked to water demand. In fact, a number of previous papers have explored the link between energy prices and water consumption (Escrivá-Bou et al. 2015; Hansen, 1996). Moreover, Section 1.5. shows that this particular aggregation choice does not affect the elasticity estimates and results are robust to different aggregations of goods. Although it would be desirable to study some other goods, such as bottled water, as a distinct category, the available price information is unfortunately not sufficiently disaggregated.

As for the demographic variables influencing demand, we control for numerous household characteristics that have been found to affect consumer behavior, particularly in terms of demand for water. Those include demographic and socioeconomic characteristics such as the size of the household (nmembers), number

[14] Unfortunately, given that the municipality code is hidden in order to guarantee the anonymity of the households, this information is not as precise as one would like in order to determine the marginal price for water paid by the household.

[15] In water demand estimation, to our knowledge only Hajispyrou et al. (2002) have used the QUAIDS functional form, but they do not estimate demand as a system, so it cannot be cited as reference here.

[16] It should be noted that, opposite to other surveys, in Spanish Consumer Expenditure Surveys expenditures on water supply are separated from expenditures on water sanitation or wastewater services, which allows us to calculate the average price for this service without the usual measurement error derived from the joint consideration of both services.

[17] This category includes electricity, natural gas, Liquefied Petroleum Gas (LPG), solid fuels, heating and central hot water consumed within the household.

[18] Some of the items included in this category are house and garage rent, house maintenance or waste and wastewater services.

of children aged between 0 and 4 years old (*nchildren1*) and between 5 and 15 (*nchildren2*), age of the household head (*age*) and dummies for the household head's level of studies—basic or no education at all (*BasicEduc*); lower secondary education (*SecondEduc1*); upper secondary education (*SecondEduc2*); or higher education (*HigherEduc*).

Some characteristics of the household residence that may influence demand for the differ-ent composites are also incorporated into the model, such as the number of rooms (*nrooms*) and size of the dwelling in square meters (*superf*), whether the household has hot water (*hotwater*)[19], type of tenancy or ownership-owned house with financial obligations (*Tenreg1*), owned without financial obligations (*Tenreg2*), rented *(Tenreg3*), rented with some allowance (*Tenreg4*), cession for low payment (*Tenreg5*) and cession free of payment (*Tenreg6*)-, type of building-detached house (*buildingtype1*), semi-detached house (*buildingtype2*), apartment in a building with fewer than 10 households (*buildingtype3)*, apartment in a building with 10 or more households (*buildingtype4*), other types of dwellings (*buildingtype5*)-, location of the household -rural (*rural*), village (*village*) or urban (*urban*)[20] - and a dummy indicating whether the residence is under 25 years old (*Hhold.lessthan25*). Finally, as in other studies (see for example Labandeira et al., 2006), we include interactions between both level of studies and household location and the variable total expenditure, as well as with year dummies, in order to control for any trends that may affect the demand for the different expenditure groups.

To estimate our model, we consider only the expenses incurred in the primary residence[21]. In order to avoid measurement errors and outliers we exclude observations with zero expenditures on food and beverage and on water, as well as those households reporting no water consumption in unit terms (zero cubic meters). Moreover, we eliminate the top and bottom 1% of the distribution of water consumed in cubic meters, unit prices and expenditures in the four groups considered.

Table 1 shows the main descriptive statistics before and after data cleaning. We can observe that averages have not changed substantially, however, the data

[19] This may have a particular effect on the relationship between price for energy and water demand that we explore in our model.

[20] We use common thresholds to distinguish between rural (fewer than 10, 000 inhabitants), village (between 10,000 and 50,000) and cities (more than 50,000).

[21] As noted by Labandeira et al. (2006), including homes other than the primary residence may lead to distortions stemming from contract overheads.

cleaning process seems to have succeeded in eliminating extreme implausible values (see Table 1) and zero expenditures. Such values can arise from several sources, including non-participation, corner solutions, infrequency of purchase or misreporting. Given that expenditures on durables, which are purchased less frequently, are excluded from the analysis and all the figures are expressed on an annual basis, zero expenditure in our sample is more likely to be related to misreporting than to other sources[22]. In any case, Table 2 shows that the percentage of households exhibiting zero expenditure in each group of commodities is extremely low, so excluding them should not lead to bias in the estimates.

As depicted in Table 1, average household consumption in Spain is 122m3 per year for the period under study, with an average price of 1.205 per m3 [23]. As is usually the case in developed countries, the water bill represents only a small fraction of family expenditure, amounting to around 0.6% of the total[24]. Figure 1 shows the distribution of residential water consumption in Spain for the period 2006-2012. As we can observe, water demand in our sample is left skewed, a shape that has also been reported in other studies (see for instance Pérez-Urdiales et al. (2015)).

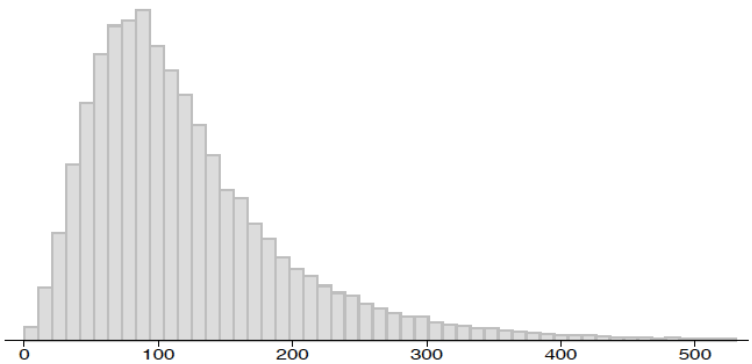


Figure 1: Distribution of annual water consumption in Spain 2006-2012.

[22] Recall that water, energy and food are basic needs, and household expenses and other expenses are rather broad categories. Therefore, non-participation, corner solutions and infrequency of purchase can be reasonably ruled out.

[23] Total water supply bill divided by the number of cubic meters consumed by the household.

[24] This is in line with other studies like Olmstead et al. (2007) which reports a budget share of 0.5%.

Table 1: Descriptive statistics before and after the data cleaning process.

	Before				After			
	N=151,068				N=133,729			
VARIABLES	Mean	SD	Min	Max	Mean	SD	Min	Max
Household income	23,912	15,807	0	386,364	23,667	14,690	0	304,488
Total expenditure (non-durables)	27,026	15,649	248.2	406,300	26,293	13,075	4,299	86,230
Total expenditure on food	5,252	3,452	0	253,578	5,149	2,823	386.9	15,645
Total expenditure on water	142.0	122.8	0	4,623	139.3	91.05	0.141	577.5
Total expenditure on energy	1,057	792.5	0	36,836	1,022	645.0	12.12	3,815
Total exp. rest of goods	12,357	10,888	0	377,223	5,149	2,823	386.9	15,645
Total exp.other household exp.	8,218	5,333	0	118,965	11,904	9,152	0	63,026
Water demand in m^3	124.5	150.1	0	33,996	122.0	77.86	0.100	530
Average price for water	1.168	0.639	0	62.62	1.205	0.433	0.0650	2.466
Nmembers	2.826	1.279	1	18	2.824	1.243	1	18
Nchildren1	0.487	0.810	0	11	0.491	0.805	0	10
Nchildren2	0.519	0.845	0	12	0.521	0.837	0	10
Age	53.61	15.26	16	85	53.48	15.17	16	85
Nrooms	5.175	1.237	0	8	5.154	1.197	1	8
Rural	0.251	0.434	0	1	0.239	0.426	0	1
Village	0.259	0.438	0	1	0.260	0.439	0	1
Cities	0.490	0.500	0	1	0.501	0.500	0	1
Basic Educ	0.276	0.447	0	1	0.270	0.444	0	1
SecondEduc1	0.305	0.460	0	1	0.309	0.462	0	1
SecondEduc2	0.161	0.368	0	1	0.164	0.370	0	1

HigherEduc	0.258	0.437	0	1	0.257	0.437	0	1
Tenreg1	0.527	0.499	0	1	0.530	0.499	0	1
Tenreg2	0.309	0.462	0	1	0.322	0.467	0	1
Tenreg3	0.0997	0.300	0	1	0.0944	0.292	0	1
Tenreg4	0.0124	0.110	0	1	0.0104	0.101	0	1
Tenreg5	0.0255	0.158	0	1	0.0253	0.157	0	1
Tenreg6	0.0258	0.159	0	1	0.0178	0.132	0	1
Buildingtype1	0.117	0.321	0	1	0.0942	0.292	0	1
Buildingtype2	0.248	0.432	0	1	0.248	0.432	0	1
Buildingtype3	0.182	0.386	0	1	0.185	0.388	0	1
Buildingtype4	0.451	0.498	0	1	0.472	0.499	0	1
Buildingtype5	0.00196	0.0442	0	1	0.00126	0.0354	0	1
Hhold.lessthan25	0.365	0.481	0	1	0.367	0.482	0	1
Hotwater1	0.996	0.0648	0	1	0.997	0.0514	0	1
BasicEduc*expenditure	5,513	10,943	0	327,071	5,375	10,366	0	82,215
SecondEduc1*expenditure	7,791	13,871	0	406,300	7,746	13,218	0	84,955
SecondEduc2*expenditure	4,724	12,313	0	238,674	4,665	11,698	0	85,134
HigherEduc*expenditure	8,998	17,810	0	307,261	8,507	16,131	0	86,230
Rural*expenditure	6,242	13,163	0	406,300	5,783	12,010	0	82,950
Village*expenditure	6,946	13,994	0	214,682	6,793	13,162	0	84,407
Cities*expenditure	13,838	18,097	0	327,071	13,716	16,646	0	86,230
Budget share for water	0.00640	0.00618	0	0.169	0.00634	0.00508	$6.96e^{06}$	0.08076

Table 2: Number and percentage of zeros in the dependent variables.

Variables	Zeros	Percentage
Food	455	0.3%
Water	5791	3.8%
Energy	1314	0.87%
Other household expenses	196	0.1%
Other non-durable commodities	10	0.001%

Figure 2[25] shows the relationship between expenditure shares and the natural log of expenditure for the composite commodities included in the analysis. As often reported in the literature (Arbués et al., 2003), the expenditure share devoted to water decreases with income. A preliminary assessment of the shape of the Engel curve for residential water demand (see Figure 2) also reveals some curvature. However, divergence from non-linear behavior is not entirely evident, so further examination is needed to determine whether higher income terms are necessary to fully capture the shape of the Engel curve for this particular good. As for the rest of the composites, budget shares for food and beverage, energy and all other goods exhibit clear non-linear behavior with respect to total expenditure. With respect to other household expenditures, this behavior is less apparent at first glance. It seems that for lower levels of income, as households become richer, they can afford more (or better quality) food, beverage, tobacco and energy. After a certain income threshold at which needs related to these goods are satisfied, a smaller share is spent on those goods and an increased share on other household expenditures, (e.g. renting a bigger or better located house).

[25] Non-parametric kernel regression. We use an Epanechnikov kernel. The bandwidth is chosen, according to the ROT method of bandwidth selection, as the one that minimizes the conditional weighted mean integrated squared error. The shape of the curve does not vary substantially depending on the choice of kernels, polynomial smooth options or bandwidths.

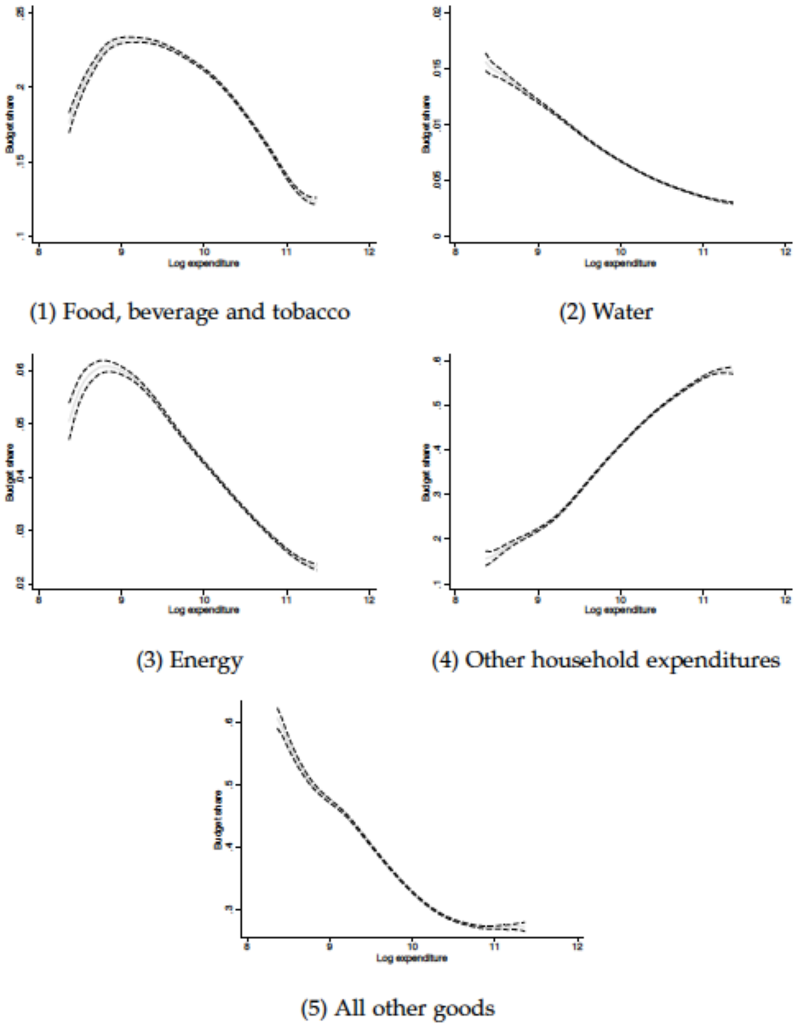


Figure 2: Non-parametric Engel curves (solid gray line) and 95% confidence intervals (dashed lines).

1.4. Empirical Analysis

1.4.1. Methodology

As explained in previous sections, we estimate a QUAIDS model (see Banks et al. (1997) for a comprehensive description). Specifically, we assume a two-step budgeting procedure as described in section 1.3.1. and use the following aggregation of non-durable goods in the second step: 1) food and beverage, 2) water, 3) energy, 4) household expenses other than water and energy and 5) all other goods. When goods are aggregated to create a single composite commodity, the Hicks-Leontief Composite Commodity Theorem (Hicks, 1939; Leontief, 1936) asserts that prices of the goods in the group should move proportionately. Typically, this assumption does not hold in most aggregations, so in order to avoid problems derived from the Composite Commodity Theorem not holding in our aggregation, we decided to use household-specific price indexes as proposed by Pashardes et al. (2014).

$$\ln p_{ih} = \sum_{k=1}^{N}(w_{ikh} - \overline{w_{ik}}) \ln r_{ik} + 0.5 \sum_{k=1}^{N}(\overline{w_{ik}} - \overline{w_{0ik}}) \ln r_{ik} \tag{11}$$

where w_{ikh} is the share of the total budget spent by a particular household *h* on each component *k* of composite commodity *i* in the current period; w_{ik} and $\overline{w_{0ik}}$ are average shares of each component *k* within the composite *i* to which it belongs for the current and base period, respectively; and *lnrik* is the natural logarithm of the price of this good *k* in the current period. The first term $\sum_{k=1}^{N}(w_{ikh} - \overline{w_{ik}})\, ln\, r_{ik}$ takes into account differences in preferences across households, while the second term is a Tornqvist index that captures the substitution effects within the composite commodity.

There are two advantages to using these indexes over fixed price indexes such as Laspeyres or Paasche. First, substitutions between the items included within each composite are accounted for, as budget shares are allowed to vary with the relative prices of those goods, thus making them valid even if the Composite Commodity Theorem is not satisfied in the aggregation. Moreover, demand patterns can be better identified, as price indexes are specific for each household (they vary according to the specific preferences of each household) and preference heterogeneity across households is incorporated into the price index. See Pashardes et al. (2014) for more details.

Data on prices used to compute the household-specific price indexes are obtained from the Consumer Price Index statistics published by the Spanish National Institute of Statistics. These indexes are disaggregated at the level of the 17 Spanish autonomous communities and 2 autonomous cities (Ceuta and Melilla) and for each year considered in the sample. Therefore, they contain temporal as well as regional variability. Moreover, instead of using a single deflator for each of the composite commodities, we have used deflators for the different components included in each composite. Since each household consumes different proportions of those component goods and we have deflators for each, applying price indexes such as those in equation (1) allows us to account for individual preferences, thus increasing variability across households. With respect to prices for water, Consumer Price Index statistics are not sufficiently disaggregated to isolate changes in prices of this particular item. However, since the EPF contains information on cubic meters of water consumed by the household, price indexes for water are built using unit or average prices derived from the sample[26]. It should be noted that, although this is standard practice, the fact that unit prices not only reflect variation in market prices, but also depend on the quality of the goods chosen by the household (and therefore, on consumer tastes), may represent a source of endogeneity (Deaton, 1988). However, this is not expected to be an issue in our case: given that the residential water supply sector in Spain is a local natural monopoly, the choice of supplier is limited and therefore households cannot choose the quality of their tap water.

Estimations are performed using the iterative linear least-squares estimator (ILLE) proposed by Blundell and Robin (1999). Although they belong to the non-linear group of demand models, QUAIDS (and AIDS) parameters are linear conditional on a series of functions of the parameters themselves and a set of explanatory variables. Exploiting this conditional linearity allows consistent estimates of the model parameters using iterations of linear moment estimators such as Seemingly Unrelated Regression (SUR) or Ordinary Least Squares (OLS) until convergence is achieved[27]. The resulting estimator has been shown to be asymptotically equivalent to the non-linear three-stage estimator (NL3S), but it offers the advantage of being more computationally efficient (see Blundell and Robin (1999) for further clarification). In order to avoid a singular error variance-

[26] Information on the utility serving the household is missing from our database, meaning that only average price can be used. However, in our opinion it is unlikely that this represents a source of misspecification. In Spain, utilities must publish new tariffs in an official gazette, either at the provincial or autonomous community level; they are not, however, obliged to publish this information either on their website or in the invoice sent to consumers, both of which are far more accessible than official gazettes. Finding out the tariffs may therefore require thorough examination of the information about water prices published in the official gazettes, a task that entails a great deal of time and effort on the part of the consumer. As opposed to marginal price, average price can be easily computed. In this case, only the total billing amount and quantity consumed is needed -information that can be easily found in a standard water bill-. Moreover, the only study conducted in Spain that contains information on consumer knowledge of water tariffs reports that most admit to not knowing the rates they pay (Pérez-Urdiales et al., 2015).

[27] In a first stage in the estimation, a Stone Price index is used as an initial value for a(p), and b(p) is assumed to be the unit vector.

covariance matrix, one of the equations has to be dropped[28]. The adding-up restriction is automatically satisfied and both homogeneity and symmetry are imposed. Following Deaton and Muellbauer (1980b) and Banks et al. (1997), a_0 takes a value below the minimum of the natural logarithm of total expenditure in our sample. However, as shown in Section 1.5., results are robust to the choice of different values for a_0.

Another issue that must be addressed is endogeneity in both total expenditure and price for water. Endogeneity of total expenditure may arise from measurement errors or the fact that total expenditure, as the sum of the expenditures on individual purchased commodities, may be jointly determined with the expenditure shares of those commodities. Therefore, as is common practice, we use total income as an identifying instrument for total expenditure. Moreover, as explained in Section 1.1., there is also serious concern that price may be endogenous in the model. Given that our database does not include information on the utility serving the household, the methods usually employed to address price endogeneity (described in the previous section) cannot be applied. However, we can take advantage of the wide regional variability in our sample and instrument price using a jackknife instrument for average price proposed by Grafton et al. (2011) also in the context of water demand estimation under block pricing (see Appendix for further information).

In our model, as proposed by Blundell and Robin (1999), endogeneity is dealt with using a two-step technique involving a non-linear IV approach and the augmented regression techniques of Hausman (1978) and Holly and Sargan (1982). First, reduced forms of the endogenous variables in our model (that is the log total expenditures and the vector of prices for the N goods) are regressed on all the exogenous explanatory variables and the set of instruments. Independent variables in this first stage include sociodemographic variables and trend dummies, the log of prices and log total expenditure when exogenous, and, as identifying instruments, the jackknife instrument for water price and log of household income to deal with endogeneity of total expenditure.

In a second step, the vector of estimated residuals from these first reduced-form regressions ($\hat{v}_h$) are retrieved and used to "augment" the demand system estimation by including them as additional regressors in the budget share equations. In the context of this augmented regression, the SUR estimator is equivalent to the 3SLS

[28] The estimator has been shown to be invariant to the choice of the equation that is dropped (Powell, 1969) and the parameters of the left-out equation can be recovered from the adding-up restriction.

and significance tests for the components of the vector of residuals in the system equations become a test of endogeneity.

1.5. Results

Table 3 presents estimated budget shares and total non-durable expenditure and own-price elasticities evaluated at sample means. The estimated budget share for water is highly significant and similar to the actual observed value (see Table 1 above). The own-price elasticity of -0.948 and income (budget) elasticity of 0.108 are also found to be significant and their magnitude is in line with the range of values previously found in the literature (Worthington and Hoffman, 2008; Arbués et al., 2003). Therefore, according to our results, water is a normal good and a necessity, and demand is found to be relatively price inelastic. It should be noted that the elasticities reported here are long-run elasticities, as the seven year time span considered in the study is long enough to allow some adjustments in durable equipment (such as dishwashers, washing machines and other water-saving appliances) in response to a change in water price. With respect to the rest of the composite goods, food, energy and other household expenditures are normal goods and necessities, while the aggregate of all other goods is found to be a luxury. Those estimates are consistent with typical values found in related studies[29].

[29] See for example the seminal paper by Banks et al. (1997), Brännlund and Nordström (2004) and Tiezzi (2005) for the case of energy estimation with similar grouping approach, and Andreyeva et al. (2010) for a review on food demand elasticities.

Table 3: Estimated budget shares and total non-durable expenditure and own-price elasticities for the groups in the demand system at sample means.

	Budget shares	Total non-durable expenditure elasticities	Conditional uncompensated own-price elasticities
Food	0.222***	0.599***	-1.162***
	(0.000)	(0.013)	(0.024)
Water	0.006***	0.108**	-0.948***
	(0.000)	(0.049)	(0.031)
Energy	0.044***	0.577***	-2.143***
	(0.000)	(0.022)	(0.017)
All other of non-durable household expenses	0.324***	0.810***	-1.021***
	(0.000)	(0.011)	(0.022)
All other non-durable commodities	0.404***	1.431***	-0.789***
	(0.000)	(0.010)	(0.017)

Standard errors in parentheses

Notes: *, ** and *** denote significance at 10%, 5%, and 1% significance levels, respectively.

Table 4 and Table 5 report cross-price elasticities[30] and the main results relating to the income terms, respectively. The high significance of most cross-price elasticities in the model implies that substitution patterns do exist in our sample. Particularly in relation to water, Table 4 shows that water demand is affected by changes in the price of all other non-durable composite goods purchased by the household, as opposed to what was indicated by previous models. Although, as already proposed in some studies (Arbués et al., 2003), water has no good substitutes among non-durable commodities, it is found to be a substitute for food and beverage and all other non-durable commodities. Contrary to previous studies on the issue by Hansen (1996) and Escrivá-Bou et al. (2015) that found a negative, although small, cross-price elasticity, energy is found to be a substitute for water. In Spain it is not

[30] A positive cross-price elasticity between two goods implies that an increase in the price of one of them leads to an increase in the demand for the other good (substitute goods), while the opposite is true for complementary goods

uncommon for households in an apartment block to have central hot water[31] provided by the building's residents' association. Central hot water comprises a mixture of expenditure on both water and energy, but as they are jointly included in the residents' association invoice issued to the household, it is not possible to isolate the expenditure on water. Since the energy to heat the water is usually much costlier than the water itself, in Consumer Budget Surveys this good is usually included in the energy category, which may explain why they appear as substitutes. Finally, as expected, expenditure on tap water is complementary to other household expenditures[32].

Table 4: Uncompensated cross-price elasticities at sample means.

	Food and beverage	Water	Energy	Other household expenditures	All other non-durable commodities
	-1.162***	0.010	0.054***	0.526***	-0.023
Food and beverage	(0.024)	(0.017)	(0.011)	(0.026)	(0.023)
Water	0.483***	-0.948***	0.344***	-0.437***	0.446***
	(0.050)	(0.031)	(0.022)	(0.051)	(0.053)
Energy	0.277***	0.044	-2.143***	1.269***	-0.023
	(0.037)	(0.025)	(0.017)	(0.039)	(0.035)
All other non-durable household expenses	0.311***	-0.012	0.162***	-1.021***	-0.252***
	(0.021)	(0.014)	(0.009)	(0.022)	(0.020)
All other non-durable commodities	-0.198***	-0.001	-0.040***	-0.403***	-0.789***
	(0.018)	(0.012)	(0.008)	(0.019)	(0.017)

Standard errors in parentheses

Notes: * , ** and *** denote significance at 10%, 5%, and 1% significance levels, respectively.

[31] It should be noted that besides affecting the relationship between water and energy consumption, this fact may further entail a source of measurement error in water consumption for these households. Unfortunately, we do not have information in the sample on which households rely on central hot water as to correct for it.

[32] Recall that some of the items included in this category are house and garage rent, household maintenance or waste and wastewater services.

With respect to the income terms, Table 5 shows that both the log of total expenditure and its square are highly significant in the budget share equations. Thus, the quadratic term is needed in all the estimated equations and the non-linear behavior with respect to income first identified in the kernel regressions depicted in Figure 2 is confirmed. This departure from linearity in the shape of the Engel curve also applies for the case of water and other household expenditures, whose kernel representations were not so clear cut. The Wald test of joint significance of the quadratic income term across the system reinforces the need to include this term (see Table 5), supporting the use of the QUAIDS specification over the AIDS model (in which similar assumptions are made, but the quadratic income term is not considered).

Table 5: Estimated budget or income effects

	Ln of total expenditure	Ln of total expenditure squared
Food	-.0879***	-.0432***
	(.0030)	(.0009)
Water	-.0054***	.0012***
	.0002	(.0000)
Energy	-.0186***	-.0012***
	(.0010)	(.0003)
All other non-durable	-.0621***	.0285***
household expenses	(.0037)	(.0011)
All other non-durable	.1740***	.01472***
commodities	(.0040)	(.0012)
Joint significance	χ^2_4 = 3383.81	p-value = 0.0000

Standard errors in parentheses.
Notes: * , ** and *** denote significance at 10%, 5%, and 1% significance levels, respectively.

With respect to endogeneity correction, Table 6 shows reduced-form estimates of the endogenous regressors in the model (i.e. total expenditure and price for water). Table 6 displays a highly significant partial correlation between total income and total expenditure, implying that it represents a suitable instrument. As presented in Table 6, price indexes for water also exhibit highly significant partial correlation with the jackknife-type variable intended to instrument it.

Table 6: Reduced-form parameters for ln of total expenditure and log of water prices

Standard errors are reported in parenthesis

	Log total expenditure	Log of water prices
Log of food prices	-.0019***	-.0001**
	(0.0002)	(0.0000)
Jackknife instrument for water price	-.0017***	.0100***
	(0.0001)	(0.0000)
Log of Energy prices	-.0017***	.0004***
	(0.0000)	(0.0000)
Log of price for other household expenses	-.0011***	-.0012***
	(0.0002)	(0.0001)
Log of price for all other goods	.0030***	-.0007***
	(0.0001)	(0.0000)
Ln of total income	.1708***	-.0013***
	(0.0014)	(0.0004)

Although they are not reported here for the sake of simplicity, 33 demographic variables and trend dummies were also accounted for. Recall that one of each type was excluded from the analysis to avoid perfect collinearity.

Notes: * , ** and *** denote significance at 10%, 5%, and 1% significance levels, respectively.

As noted in Section 1.4.1., the tests for significance of the coefficients for the components of the vector of estimated residual ($\hat{v}_h$) from the reduced-form regressions of the endogenous regressors in the augmented system become tests for endogeneity. Tests for the endogeneity of total expenditure are reported in Table 7. Exogeneity is rejected for all the composite commodities except water. However, the need to control for expenditure endogeneity is confirmed by the rejection of the null of the joint test on log of total expenditure being exogenous across all the budget share equations (see bottom of Table 7). The test for the endogeneity of the water price in the water equation is also strongly rejected[33]. The validity of the instruments is discussed above.

Table 7: Tests for the endogeneity of total expenditure for all the equations in the demand system.

	Test for the endogeity of total expenditure
Food	123.36***
Water	0.92
Energy	23.50***
All other non-durable household expenses	40.41***
All other non-durable commodities	212.14***
Joint significance	χ^2_4 = 282.39 p-value = 0.0000

Notes: *, ** and *** denote significance at 10%, 5%, and 1% significance levels, respectively.

With respect to the demographics in our model, it can be observed (see Table 8 that many of them are significant in the water share equation. The number of rooms in the household appears negatively correlated to water demand. It also seems that rural households and those located in villages consume less water than their urban counterparts, probably because they have easier access to sources of water other than the tap, such as fountains and wells. Households in a rented residence, with (regten4) or without rental assistance (regten3), and in residences that are owned but with financial obligations (regten2), are reported to devote a higher proportion of their total non-durable expenditure to water than the ones with tenancy under a cession free of payment (reference), or those living in houses under 25 years old and with greater total area. With respect to the number of household members, this seems to be positively correlated with water demand. Conversely, the more children aged either between 0 and 5 (nmiem7) or between 5

[33] χ^2_4 =50.24. p-value =0.000.

and 15 (nmiem3), the less the proportion of total expenditure devoted to residential water consumption[34]. In a reference household in which the household head has completed only lower secondary education, household water demand seems to be smaller than for households in which any other education level has been completed. Finally, as expected, water consumption is shown to be higher for detached and semi-detached houses, which usually have more water-dependent amenities such as gardens and swimming pools.

Table 8: Parameter estimates for the demographics in the model in the water budget share equation

	Coefficient	Standard error
Nrooms	-.0002***	(0.0000)
Size (m^2)	5.71e^{-06}***	(0.0000)
Hotwater	.0008***	(0.0002)
Buildingtype1	.0008**	(0.0003)
Buildingtype2	.0009***	(0.0003)
Buildingtype3	-.00004	(0.0003)
Buildingtype4	-.0004	(0.0003)
Tenreg1	.00002	(0.0001)
Tenreg2	.0003***	(0.0001)
Tenreg3	.00174***	(0.0001)
Tenreg4	.0008***	(0.0002)
Tenreg5	.0001063	(0.0001)
Hhold.lessthan251	.0002***	(0.0000)
Year2006	.0006***	(0.0001)
Year2007	-.0002***	(0.0001)

[34] It has been found that the presence of individuals belonging to vulnerable populations (e.g. young children, elderly people or individuals with poor health) in the household can generate risk aversion, sometimes triggering the decision to consume bottled water (Zivin G. et al., 2011; Yoo and Yang, 2000). Therefore, it is more likely that these household are using bottled water to cover some of their water needs, such as drinking and cooking, and are thus spending a lower proportion of the household budget on tap water.

Year2008	-.0004***	(0.0001)
Year2009	.0000151	(0.0001)
Year2010	-.0001169**	(0.0001)
Year2011	-.0002***	(0.00004)
Age	$-1.56e^{-07}$	(0.0000)
Nmembers	.0008***	(0.00002)
Nchildren1	-.0001**	(0.00002)
Nchildren2	-.0002***	(0.00002)
BasicEduc	.0003***	(0.0002)
SecondEduc2	.0004**	(0.0002)
HigherEduc	-.0003**	(0.0001)
Rural	-.00392***	(0.0001)
Village	-.0007***	(0.0001)
Basic*expenditure	$-6.40e^{-09}$	(0.0000)
SecondEduc2*expenditure	$-1.11e^{-08}$*	(0.0000)
HighEduc*expenditure	$7.04e^{-09}$	(0.0000)
Rural*expenditure	$5.37e^{-08}$***	(0.0000)
Village*expenditure	$-1.05\ e^{-09}$	(0.0000)
Constant	.0041***	(0.0004)

Note: One dummy from each group of socioeconomic characteristics is excluded from the analysis in order to avoid perfect collinearity. The household considered as the reference has the following characteristics: Located in Andalucia, in a city, in 2012, without hot water, tenancy with cession free of payment, in "other type of dwellings", residence over 25 years old and household head with lower secondary education.

1.5.1. Robustness checks

Robustness checks have been performed using different aggregations of goods, different initial values for α_0 and changing the demographics included in the model. They show that our results are robust to other choices that could have been made with respect to these particular issues. Results for the price and income elasticities for residential water demand under different α_0 displayed in Table 9 show only slight variations in the price elasticity estimates. Moreover, in our model, some of the goods (energy and other household expenditures) were separated from the rest of the goods. Table 10 and Table 11 show that this particular choice does not affect estimates for cross-price elasticities.

Table 9: Price and income elasticities for residential water demand under different values of α_0

	Elasticities	
	Income	Price
α_0= 2.75	-0.108**	-0.950***
α_0= 4.125	-0.108**	-0.955***
α_0= 4.95	-0.108**	-0.948***
α_0= 5.5 (Baseline)	-0.108**	-0.948***
α_0= 6.05	-0.108**	-0.947***
α_0= 8.2	-0.108**	-0.944***

Notes: *, ** and *** denote significance at 10%, 5%, and 1% significance levels, respectively.
Elasticities are reported at sample means.
Notes: Values for a_0 included here correspond to, respectively, 10%, 20% and 50% increases and decreases from the baseline level used for the estimations.

Table 10: Uncompensated cross-price elasticities at sample means under different aggregations (I).

	Food and beverage	Water	Energy	All other non-durable commodities
Food	-1.072***	0.01	0.112***	0.367***
	(-0.024)	(-0.017)	(-0.011)	(-0.027)
Water	0.480***	-0.965***	0.281***	0.125*
	(-0.049)	(-0.031)	(-0.021)	(-0.059)
Energy	0.554***	0.035	-2.042***	0.825***
	(-0.035)	(-0.025)	(-0.016)	(-0.041)
All other commodities	-0.016*	-0.005	0.027***	-1.163***
	(-0.007)	(-0.005)	(- 0.003)	(-0.008)

Standard errors in parentheses
Notes: *, ** and *** denote significance at 10%, 5%, and 1% significance levels, respectively.

Table 11: Uncompensated cross-price elasticities at sample means under different aggregations (II).

	Food and beverage	Water	All other non-durable commodities
Food	-0.967***	0.01	0.405***
	(-0.023)	(-0.017)	(-0.024)
Water	0.487***	-0.966***	0.514***
	(- 0.049)	(-0.031)	(-0.053)
All other commodities	-0.013*	-0.003	-1.120***
	(-0.007)	(-0.005)	(-0.007)

Standard errors in parentheses
Notes: *, ** and *** denote significance at 10%, 5%, and 1% significance levels, respectively.

1.5.2. Model performance and comparison with other functional specifications

As a final step of the empirical analysis, we estimate different functional demand specifications and compare them in terms of performance. Specifically, we estimate the linear, log-linear and double-log models, which are the most common specifications in the water demand literature. Moreover, we include in the analysis the AIDS model, as it is based on similar behavioral assumptions as QUAIDS, but excluding the quadratic income term from the specification.

Table 12 reports the results of the IV regressions for the linear, log-linear and double log models, using the jackknife instrument proposed by Grafton et al. (2011) as an instrument for water price (see Section 1.4.1. for a more detailed description) and accounting for trend dummies and the 27 demographic characteristics[35] previously included in the QUAIDS[36].

[35] After excluding one dummy of each type in order to avoid perfect collinearity, 27 demographic controls are included in the final estimations.

[36] The first stage of the IV estimations are also included in *Table 14* in the Appendix. Following Bound et al. (1995), we report the R^2 of the first stage when included instruments are partialled out (Partial R^2) as well as an F-test of excluded instruments. Relevance of the jackknife-type instrument is confirmed by the high Partial R^2 of the excluded instrument and the rejection of the F-test of excluded instruments in all the estimated models. Validity is discussed in Section 1.4.1. In addition, Hausman tests for endogeneity are rejected for every estimated model, underscoring the suitability of Instrumental Variables estimation.

Table 12: IV estimates of different functional specifications for water demand.

	Water demand models		
	Linear model	Log-linear	Double-log
Average price	-46.45***	-0.378***	
	(0.470)	(0.00386)	
Income	$-7.02e^{-06}$	$-8.66e^{-09}$	
	$(1.79e^{-05})$	$(1.47e^{-07})$	
Log of average price			-0.400***
			(0.00420)
Log of income			0.00365
			(0.00376)
Controls included [a]	Yes	Yes	Yes
N	133, 598	133, 222	133, 222
R^2	0.160	0.167	0.165
Wald test of	728.31	764.83	749.45
joint significance [cb]	(0.000)	(0.000)	(0.000)
Hausman test[c]	7, 897.67	10, 416.33	11, 062.65
	(0.000)	(0.000)	(0.000)

Notes: *, ** and *** denote significance at 10%, 5%, and 1% significance levels, respectively. Elasticities are reported at sample means.
Standard errors in parentheses.
[a] The same 33 demographic variables and trend dummies included in the QUAIDS model were included in these models as controls.

bIt follows a distribution of 35 degrees of freedom [c] P-value in parentheses

Table 13 shows the price-elasticities yielded by the different model specifications. All the reported elasticities are in line with the values reported in related studies (see Arbués et al. (2004) and Worthington and Hoffman (2008)), although the estimates for linear, log-linear and double-log models are slightly below the lower limit of the usual range for long-run elasticities[37]. We can observe that the price elasticity yielded by the QUAIDS and AIDS models are higher than those produced by the rest of the models. The only previous application of QUAIDS to water demand estimation (Hajispyrou et al., 2002) also reported relatively high elasticities, but that study is not directly comparable to ours, as the authors assume separability of water consumption, and thus do not estimate water demand through a system of equations but rather with a single equation. Moreover, although QUAIDS models have been widely applied in the energy literature (Pashardes et al., 2014; Labandeira et al., 2006; Tiezzi, 2005), no studies to date have attempted to make comparisons with alternative demand specifications so as to determine whether the differences found in the estimated elasticities are systematic.

Table 13: Price elasticities of demand.

	Water demand models				
	QUAIDS	AIDS	Linear	Log-linear	Double-log
Price elasticity	-0.948***	-0.906***	-0.459***	-.456***	-.4003***
N	133, 598	133, 598	133, 598	133, 222	133, 222
R^2	0.2945[a]	0.2904[a]	0.160	0.167	0.165

Notes: *, ** and *** denote significance at 10%, 5%, and 1% significance levels, respectively. Elasticities are reported at sample means.
[a]This R^2 corresponds to the water equation.

With respect to the goodness of fit, QUAIDS offers the highest R^2 of all the estimated models. Moreover, Figure 3 shows the real distribution for water demand as well as pairwise comparisons of the predicted distributions using the different estimated models and QUAIDS. We can see that QUAIDS is not only closer to the

[37] The usual reported range for these elasticities is between -0.5 to -1 (Worthington and Hoffman, 2008).

observed data than the other functional specifications (implying better overall fit), but it also shows a better fit to the asymmetric shape of the real distribution.

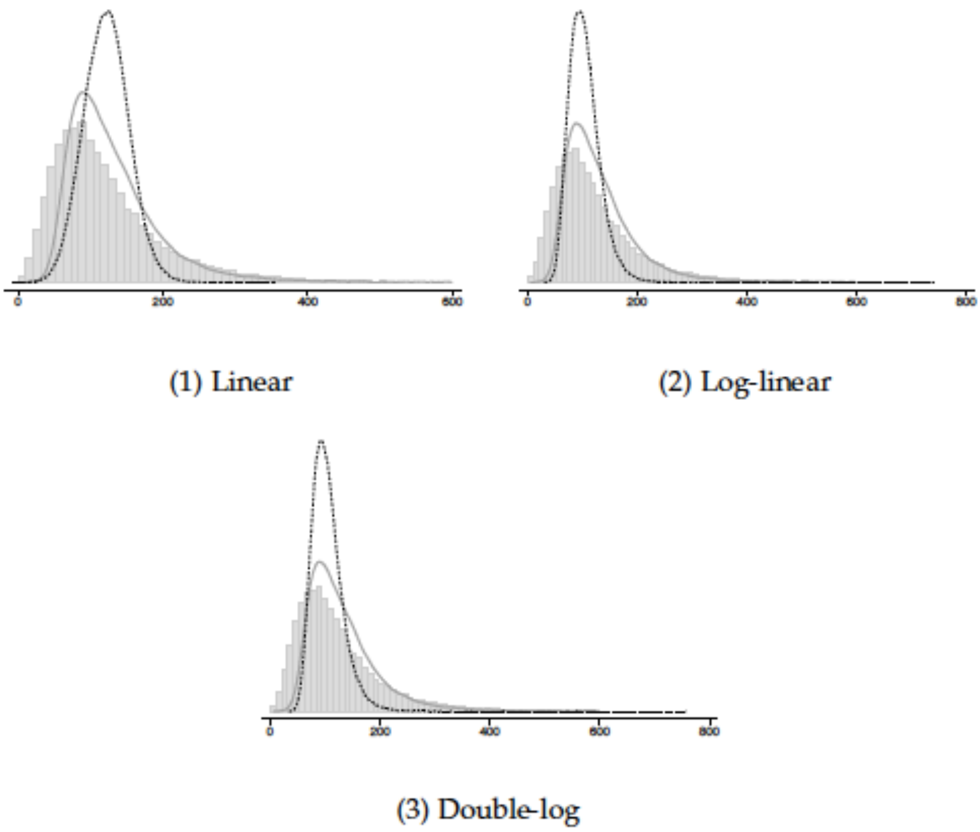


Figure 3: Distribution of annual residential water demand (m3) in Spain 2006-2012 (histogram). Distribution of the predicted value using QUAIDS model (solid line) and several model specifications (dashed line).

1.6. Conclusions

Accurate estimates of demand patterns are crucial to the implementation of policies that adequately address growing water scarcity and help to achieve a better management of water resources. Consequently, the related body of literature is extensive and covers a significant number of issues and approaches. Several theoretical specifications have been used in demand estimation, with the linear, log-linear and double log being the most common. However, despite the considerable attention that this subject has attracted, there has been little debate on which specification is best in terms of fit and performance (European Commission, 2015). Most estimations of income and price elasticities for residential water are based on single-equation models that rely on assumptions of weak separability of water consumption from other goods (i.e. assuming that household water consumption does not depend on the price of other goods) and linear income effects on water consumption.

In this paper, we relax these assumptions by using a less restrictive system of demand estimation, the Quadratic Almost Ideal Demand System (QUAIDS), and we compare it with the alternative functional forms most commonly used in the literature. Employing a rotating panel of 151.068 observations representative of the Spanish population for the period 2006 2012, we find that the long-run price and income elasticities yielded by the QUAIDS model are -0.974 and 0.108, respectively, indicating that water is a normal good, a necessity and relatively price inelastic.

Moreover, this study reveals the existence in our sample of non-linearities in Engel curves for residential water demand, as well as substitution and complementary patterns deriving from changes in prices relative to other groups of commodities. In this context, the use of single-equation models such as the linear, log-linear and double-log specifications seems to impose implausible restrictions on water demand. Conversely, QUAIDS can be said to offer a richer picture of consumer expenditure patterns, as it is shown to be more consistent with consumer behavior. The higher R2 reported for this model as opposed to single-equation models and the fact that, as shown in Figure 3, QUAIDS models are both closer to the observed data than the other specifications and shows a better fit to the asymmetric shape of the real distribution for water demand, would seem to confirm our supposition.

Therefore, our results suggest that, in order to avoid bias in the estimates, and when the necessary information is available, water demand should be estimated in relation to the demand for the rest of the goods consumed in the household and non-linear behavior with respect to income should also be accounted for. Moreover, our findings call for a more in-depth debate as to the preferred form of estimation, in order to provide policy makers with models that better reflect consumer behavior and that are more consistent with utility theory. According to our study, QUAIDS could provide a better understanding of water demand than previously used models. With the increasing availability of detailed microdatasets on household consumption and expenditures, the model proposed in this paper could be used to further exploit the information contained in those databases, leading to more accurate estimates of the expected impact of water management policies. In addition, given that Consumer Expenditure Surveys comparable to those used in this paper are available in many countries, similar analyses could be conducted to predict national and regional demand, thus helping to address the future challenges posed by water scarcity.

Appendix

In Grafton et al. (2011), the jackknife grouping approach by Angrist et al. (1999) is used to obtain an instrument for price. The instrument for household h is computed as follows:

$$z_h \equiv \frac{N\,\overline{\ln p_{\imath h}} - ln\ p_{ih}}{N-1} \tag{12}$$

where $\overline{\ln p_{\imath h}}$ is the mean of the water price indexes for households within a particular group lnp or cluster of households, and N is the number of households within each group.

To define the clusters, households are classified according to several location characteristics described in the database that have been found to affect water prices (Martínez-Espiñeira et al., 2009) including population density, range of total population, region (i.e. Spanish Autonomous Communities), and whether the municipality is the capital of the province in which it is located. Note that the instrument takes different values for each household and the computation excludes the price paid by that same household, considering only the average of the rest of the households belonging to the same group. As argued by Grafton et al. (2011), the average price for all households other than the one considered each time is by definition uncorrelated with any endogenous consumption decision taken by the household, thus making this ’leave-one-out’ jackknife-type instrument a valid instrument for price. Moreover, the characteristics determining the grouping approach are supply-side determinants of price, and are therefore not expected to be correlated to water demand.

Table 14: First stage of the IV estimates and related test statistics.

	Water demand models		
	Linear model	Log-linear	Double-log
Income	-5.91e-08	-5.91e-08	
	(8.55e-08)	(8.55e-08)	
Jackknife-type instrument	0.998***	0.998***	
	(0.00416)	(0.00416)	
Log of income			-0.00171
			(0.00207)
Jackknife-type instrument for log of price			0.997***
			(0.00495)
Controls included [a]	Yes	Yes	Yes
N	133, 598	133, 598	133, 222
F-test of joint significance [b]	2057.41	2052.74	11062.65
	(0.000)	(0.000)	(0.000)
R^2	0.3503	0.3504	0.3479
Partial R^2	0.2926	0.2925	0.165
F-test of excluded instruments[ab]	55236.7	55069.4	54376.67
	(0.000)	(0.000)	(0.000)

Notes: *, ** and *** denote significance at 10%, 5%, and 1% significance levels, respectively. Elasticities are reported at sample means. Standard errors in parentheses.

[a] Follows a distribution of 35 degrees of freedom

[b] P-value in parenthes

References

AEAS-AGA (2012). XII Edición "Encuesta de Suministro de Agua Potable y Saneamiento en España ".Can be accessed in: http://www.asoaeas.com/sites/default/files/Documentos/Nota

AEAS-AGA (2013). Estudio AEAS-AGA 2013 - El agua en España. Madrid, Spain.

Al-Quanibet, M. H. and Johnston, R. S. (1985). Municipal demand for water in Kuwait: methodological issues and empirical results. Water Resources Research, 2:433–8.

Andreyeva, T., Long, M. W., and Brownell, K. D. (2010). The impact of food prices on consumption: A systematic review of research on the price elasticity of demand for food. American Journal of Public Health, 100(2):216–222.

Angrist, J. D., Imbens, G. W., and Krueger, A. B. (1999). Jackknife instrumental variables estimation. Journal of Applied Econometrics, 14(1):57–67.

Arbués, F., Barberán, R., and Villanúa, I. (2004). Price impact on urban residential water demand: A dynamic panel data approach. Water Resources Research, 40(11).

Arbués, F., García-Valiñas, M. Á., and Martínez-Espiñeira, R. (2003). Estimation of residential water demand: A state-of-the-art review. Journal of Socio-Economic, 32:81–102.

Arnell, N. (1999). Climate change and global water resources. Global Environmental Change, 9:S31–S49.

Arnell, N. (2000). Impact of climate change on global water resources: Volume 2, unmitigated emis-sions. Report to Department of the Environment, Transport and the Regions. University of Southampton, Southampton, United Kingdom, 53 pp.

Atkinson, A., Gomulka, B. J., and Stern, N. H. (1990). Spending on alcohol: Evidence from the family expenditure survey 19701983. Economic Journal, 100:808–827.

Baerenklau, K. A., Schwabe, K., and Dinar, A. (2014). The residential water demand effect of increasing block rate water budgets. Land Economics, 90(4):683–699.

Banks, J., Blundell, R., and Lewbel, A. (1997). Quadratic engel curves and consumer demand. Review of Economics and Statistics, 79:527–539.

Biswas, A. K. and Tortajada, C. (2009). Water Management in 2020 and Beyond, chapter Changing Global Water Management Landscape, pages 1–33. Springer.

Blundell, R. (1988). Consumer behaviour: Theory and empirical evidence. A survey. The Economic Journal, 98:16–65.

Blundell, R., Pashardes, W. P., and Weber, G. (1993). What do we learn about consumer demand patterns from micro-data? American Economic Review, 83:570–597.

Blundell, R. and Robin, J.-M. (1999). Estimation in large and disaggregated demand systems: An estimator for conditionally linear systems. Journal of Applied Econometrics, 14:209–232.

Borenstein, S. (2009). To what price do consumers respond? Residential demand elasiticity under increasing-block pricing. http://faculty.haas.berkeley.edu/borenste/download/NBERSI2009.pdf (accessedOctober24, 2015).

Bound, J., Jaeger, D. A., and Baker, R. M. (1995). Problems with instrumental variables es-timation when the correlation between the instruments and the endogeneous explanatory variable is weak. Journal of the American Statistical Association, 90(430): 443–450.

Brännlund, R. and Nordström, J. (2004). Carbon tax simulations using a household demand model. European Economic Review, 48:211–233.

Dalhuisen, J., Florax, R., Groot, H., and Nijkamp, P. (2003). Price and income elasticities of residential water demand: a meta-analysis. Land Economics, 79(2):292–308.

De Bartolome, C. (1995). Which tax rate do people use: Average or marginal? Journal of Public Economics, 56(1):79–96.

Deaton, A. (1988). Quality, quantity and spatial variation of price. American Economic Review, 78:418–430.

Deaton, A. and Muellbauer, J. (1980a). An Almost Ideal Demand System. The American Eco-nomic Review, 70(3):312–326.

Deaton, A. and Muellbauer, J. (1980b). Economics and Consumer Behavior. Cambridge University Press. Cambridge, UK.

Deaton, A. and Zaidi, S. (2002). Guidelines for constructing consumption aggregates for welfare analysis. LSMS Working Paper; No. 135. World Bank. https://openknowledge.worldbank.org/handle/10986/14101.

Dharmaratna, D. and Harris, E. (2010). Estimating residential water demand using the Stone-Geary functional form: the case of Sri Lanka. Discussion paper, 46/10, Department of Economics, Monash University.

Engel, E. (1895). Die lebenskosten belgischer arbeiter-familien fruher and jetzt. International Statistical Institute Bulletin, 9:1–74.

Espey, M., Espey, J., and Shaw, W. D. (1997). Price elasticity of residential demand for water: A meta-analysis. Water Resources Research, 33(6):1369–1374.

European Commission (2015). Modelling household water demand in Europe. Insights from a cross-country econometric analysis of EU-28 countries joint research center technical report. Joint Research Center. Institute of Environment and Sustainability. Ispra (VA), Italy, 2015.

Escriva-Bou, A., Lund, J. R., and Pulido-Velazquez, M. (2015), Optimal residential water conservation strategies considering related energy in California. Water Resources Research, 51: 4482–4498, doi:10.1002/2014WR016821.

Gaudin, S., Griffin, R., and Sickes, R. (2001). Demand specification for municipal water management: Evaluation of the Stone-Geary form. Land Economics, 77(3):399–422.

Grafton, R. Q., Ward, M. B., To, H., and Kompas, T. (2011). Determinants of residential water consumption: Evidence and analysis from a ten country household survey. Water Resources Research, 47.

Hajispyrou, S., Koundouri, P., and Pashardes, P. (2002). Household demand and welfare: implications of water pricing in Cyprus. Environment and Development Economics, 4:659 – 685

Hansen, L. G. (1996). Water and energy price impacts on residential water demand in Copenhagen. Land Economics, 72(1):66–79.

Hausman, J. A. (1978). Specification tests in econometrics. Econometrica, 46:1251–1271.

Hausman, J. A., Newey, W. K., and Powell, J. L. (1995). Nonlinear errors in variables: Estimation of some Engel Curves. Journal of Econometrics, 65:205–233.

Hewitt, J. and Hanemann, W. (1995). A discrete/continuous choice approach to residential water demand under block rate pricing. Land Economics, 71:173–192.

Hicks, J. (1939).Value and Capital: An Inquiry into Some Fundamental Principles of Economic Theory. Oxford: Clarendon Press. Oxford.

Hildenbrand, W. (1994). Market Demand: Theory and Empirical Evidence. Princeton University Press. Princeton, New Jersey.

Holly, A. and Sargan, D. (1982). Testing for exogeneity in a limited information framework. Cahiers de Recherches 8204, Universit Lausanne.

INE (2006-2012). Encuesta de presupuestos familiares. Madrid. Spain.

IPCC (2014). Climate change 2014: Synthesis report. contribution of working groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [core writing team, R.K. Pachauri and L.A. Meyer (eds.)]. Technical report, IPCC, Geneva, Switzerland.

Ito, K. (2014). Do consumers respond to marginal or average price?. Evidence from nonlinear electricity pricing. American Economic Review, 104:537–563.

Labandeira, X., Labeaga, J., and Rodrez, M. (2006). A residential energy demand system for Spain. Energy Journal, 27(2):87–111.

Leontief, W. (1936). Composite commodities and the problem of index numbers. Econometrica, 4:39–59.

Lewbel, A. (1991). The rank of demand systems: Theory and nonparametric estimation. Econometrica, 59:711730.

Liebman,J. B. and Zeckhauser, R. (2004). Schmeduling. www.hks.harvard.edu/jeffreyliebman/schmeduling.pdf (accessed October 24, 2015).

Madhoo, Y. (2009). Policy and nonpolicy determinants of progressivity of block residential water rates- a case study of Mauritius. Applied Economics Letters, 16:211–215.

Martínez-Espiñeira, R., García-Valiñas, M. Á., and Gómez, F. G. (2009). Does private management of water supply services really increase prices? An empirical analysis in Spain. Urban Studies, 46:923–945.

Martínez-Espiñeira, R. and Nauges, C. (2004). Is all domestic water consumption sensitive to price control?. Applied Economics, 36:1697–1703.

Nataraj, S. and Hanneman, M. (2011). Does marginal price matter? A regression discontinuity approach to estimating water demand. Journal of Environmental Economics and Management, 61(2):198–212.

Nauges, C. and Whittington, D. (2010). Estimation of water demand in developing countries: an overview. The World Bank Research Observer, 25(2):263–294.

Nichele, V. and Robin, J.-M. (1995). Simulation of indirect tax reforms using pooled micro and macro French data. Journal of Public Economics, 56:225–244.

Nieswiadomy, M. and Molina, D. (1989). Comparing residential water estimates under decreasing and increasing block rates using household data. Land Econonomics, 65(3):280–89.

Nordin, J. (1976). A proposed modification to Taylors demand analysis comment. The Bell Journal of Economics, 7(2):719–721.

Olmstead, S. (2009). Reduced-form versus structural models of water demand under nonlinear prices. Journal of Business & Economic Statistic, 27(1):84–94.

Olmstead, S. M., Hanemann, W. M., and Stavins, R. N. (2007). Water demand under alternative price structures. Journal of Environmental Economics and Management, 54:181–198.

Olmstead, S. M. and Stavins, R. N. (2009). Comparing price and nonprice approaches to urban water conservation. Water Resources Research, 45(4).

Pashardes, P., Pashourtidou, N., and Zachariadis, T. (2014). Estimating welfare aspects of changes in energy prices from preference heterogeneity. Energy Economics, 42:58–66.

Pérez-Urdiales, M., García-Valiñas, M., and Martínez-Espiñeira, R. (2015). Responses to changes in domestic water tariff structures: a latent class analysis on household-level data from Granada, Spain. Environmental and Resources Economics, 63(1,):167–191.

Pollak, R. A. and Wales, T. J. (1978). Estimation of complete demand systems from household budget data: The linear and quadratic expenditure systems. The American Economic Review, 68(3):348–359.

Pollak, R. A. and Wales, T. J. (1981). Demographic variables in demand analysis. Econometrica, 49:1533–1551.

Powell, A. A. (1969). Aitken estimators as a tool in allocating predetermined aggregates. Journal of the American Statistical Association, 64:913–22.

Saez, E. (1999). Do taxpayers bunch at kink points? National Bureau of Economic Research , Working Paper 7366.

Saez, E. (2010). Do taxpayers bunch at kink points? American Economic Journal: Economic Policy, 2(3):180–212.

Schlosser, C., Strzepek, K., Gao, X., Gueneau, A., Fant, C., Paltsev, S., Rasheed, B., Smith-Greico, T., B., E., Jacoby, H., and Reilly, J. (2014). The future of global water stress: An integrated assessment. Technical report, Joint programme report series, MIT Joint Programme on the Science and Policy of Global Change.

Sebri, M. (2016). Forecasting urban water demand: A meta-regression analysis. Journal of Environmental Management, 183(3):777–785.

Shin, J. (1985). Perception of price when information is costly: Evidence from residential electricity demand. Review of Economics and Statistics, 67(4):591–598.

Stone, R. (1954). Linear expenditure systems and demand analysis: An application to the British demand. The Economic Journal, 64(255):511–27.

Taylor, L. D. (1975). The demand for electricity: A survey. The Bell Journal of Economics, 6:74–110.

Tiezzi, S. (2005). The welfare and distributive effects of carbon taxation on Italian households. Energy Policy, 33(4):1597–1612.

UN Population Division (2014). World urbanization prospects: The 2014 revision, highlights (st/esa/ser.a/352). Technical report, United Nations, Department of Economic and Social Affairs, Population Division.

UN Population Division (2015). World population prospects 2015. Technical report, United Nations, Department of Economic and Social Affairs, Population Division.

Wichman, C. J. (2014). Perceived price in residential water demand: Evidence from a natural experiment. Journal of Economic Behavior & Organization, 107, part A:308–323.

World Bank Group (2016). High and dry : Climate change, water, and the economy. Technical report, World Bank, Washington, DC. World Bank.

World Economic Forum (2016). The global risks report 2016. 11th edition. World Economic Forum. Cologny/Geneva. Switzerland.

Worthington, A. C. and Hoffman, M. (2008). An empirical survey of residential water demand modelling. Journal of Economics Surveys, 22(5):842–87.

Yoo, S.-H. and Yang, C.-Y. (2000). Dealing with bottled water expenditure data with zero observations. Economics Letters, 66:151157.

Zivin G., J., Neidell, M., and Schlenker, W. (2011). Water quality violations and avoidance behavior: Evidence from bottled water consumption. American Economic Review: Papers and Proceedings, 101(3):448–453.

Essay 2: A proposal for the analysis of price escalation within water tariffs. The impact of the Water Framework Directive in Spain.

Abstract

During the last few decades, numerous international organizations have emphasized the role of pricing policy as a tool to achieve objectives of efficiency, environmental sustainability, and cost-recovery in the management of water resources. Incorporating a certain level of price escalation within water tariffs by adopting increasing block rates (IBRs) is commonly advocated as a key element for controlling water demand and fulfilling these objectives. However, despite its widespread use, there exists no established procedure to measure the levels of price escalation embodied in water tariffs. We propose a measure of price escalation within water tariffs at the level of the water supply management unit (the municipality, in our study). In order to illustrate the usefulness of our measure, we analyse the evolution of price escalation in residential water tariffs between 2000 and 2014 in a sample of 952 Spanish municipalities and examine the factors influencing this evolution.

2.1. Introduction

Water scarcity is expected to be one of the most critical challenges for humankind in the near future (UN-Water 2006, p. 4). According to the World Economic Forum (WEF 2016), water crises are the 9th global risk in likelihood of occurrence and the third in terms of potential impact. Consequently, the efficient and sustainable use of water resources will become paramount in the agenda of global leaders and international organizations in the forthcoming decades. Policy development in this area aims at the formulation of pricing schemes that can simultaneously accommodate several objectives, which can be synthesized as the generation of enough revenue to achieve cost recovery, while pursuing equity, efficiency, and sustainability goals (Rogers *et al.* 2002). First, water is considered a basic need, so universal access to a certain level of consumption must be guaranteed (Bovis 2005). Among other issues, that implies the need for adopting affordable prices (Van de Walle 2008, Martins et al. 2013a, 2016). On the other hand, pricing policies are expected to play a more prominent role in helping achieve full recovery of the cost of the service and making a more efficient and sustainable use of the resource.

Reconciling multiple objectives in a single instrument represents a difficult challenge, particularly if only linear tariff structures are applied (Wichelns 2013; Farolfi and Gallego-Ayala 2014; Schoengold and Zilberman 2014). For this reason, two-part tariffs that include a fixed fee –per connection- and a variable or volumetric fee – related to the level of consumption- have become widespread (OECD 2010; GWI 2014; IWA 2014; Dinar *et al.* 2015). The fixed part of the fee is supposed to provide financial stability to the utilities by guaranteeing an amount of revenue sufficient to cover at least part of the cost of the service, while the variable component is expected to help meet the other objectives. Increasing block rates (IBRs henceforth) are usually recommended (European Commission 2000) for the latter, as a better means to reconcile the objectives of affordability, efficiency, and sustainability. Although water demand is relatively inelastic to price, there is evidence to suggest that substantial water savings can be achieved with increasing block rates (Olmstead 2007; Baerenklau *et al.* 2014). Grafton *et al.* (2011) conclude, for instance, that households that do not face increasing block prices use around a third more water than similar households in a volumetric pricing framework. Besides, water price elasticity has been found to increase with consumption, especially when it is intended for recreational purposes, such as gardening and maintaining swimming pools (European Environmental Agency 2012).

In this article, we quantify the results of efforts to enhance price escalation within water tariffs. To this end, we define a measure that permits full comparability among jurisdictions regardless of the specific tariff structure they use (i.e. regardless of the number and size of price blocks, presence and size of fixed components, and price levels included in the tariff). We also illustrate this approach by analysing empirical evidence for the Spanish case, specifically examining the evolution between 2000 and 2014 in 952 Spanish municipalities of the two indicators of price escalation[38] we propose. In addition, we discuss the environmental, political, and business factors that influence those changes in the degree of price escalation in the tariffs between 2000 and 2014. Our results allow to identify the scenarios most suitable for the implementation of changes in tariff policy, as well as typical scenarios in which regulators would be expected to face more resistance and require more effort to promote water tariff reforms. To the authors' knowledge, this is the first work that proposes an evaluation of the change of the level of price escalation of the water tariffs for residential uses.

The article is organized as follows. While Section 2.2. provides a review of the relevant literature on the influence of pricing schemes on demand, Section 2.3. outlines the construction of the proposed measure. In Section 2.4. an empirical implementation of the indicators is conducted. Finally, Section 2.5. reports and discusses our results, before our concluding in Section 2.6.

2.2. Literature review.

2.2.1. Price variable.

The way block pricing affects water demand is not straightforward. The modelling of water demand under block pricing has generated a long-standing debate during the last few decades. Consumer theory predicts that, under perfect information and faced with the piecewise budget constraint that corresponds to a block pricing structure, consumers optimise their consumption by responding to the actual marginal price structure of the tariff they confront. In fact, some papers have

[38] It should be noted that price escalation should not be understood here as the increase in time of the price level. Our notion of price escalation refers to individuals being charged a higher price per unit the more they consume, independently of their level of income.

found that consumers do respond to marginal price (Nataraj and Hanneman, 2011; Baerenklau et al., 2014). However, since under block pricing consumers do not face a single price but a complete tariff scheme, Taylor (1975) first suggested that the sole marginal price could not be enough to capture their demand decisions. Instead, he proposed using both a marginal and an average price. Building on Taylor's work, Nordin (1976) introduced what is known as Nordin's difference variable,[39] in an attempt to account for the income (or *intramarginal*) effect resulting from consumers moving from one block to another. In the 1990s, the discrete-continous choice (DCC) framework (Hewitt and Hanemman, 1995) was applied to water demand, allowing to take into account the existence of a piecewise budget constraint under a block pricing structure.

However, consumption decisions are most often made in the absence of perfect information. In fact, empirical evidence has shown that consumers are not always perfectly informed about the tariff structure (Nieswiadomy and Molina 1989; Nataraj and Hanemann 2011; Perez-Urdiales *et al.* 2015) and that the cognitive effort to understand complex pricing and non-linear structures is usually substantial (Nieswiadomy and Molina 1989; De Bartolome 1995). Random shocks to consumers' demand (Borenstein 2009) or income (Saez 2010) may be another source of suboptimizing consumption behaviour.[40] Consequently, economic theory has long attempted to provide guidance on how best to identify or proxy the price perceived by consumers under imperfect information. For instance, Saez (1999, 2010) and Borenstein (2009) relax the assumption of perfect information by allowing the existence of random shocks to income and demand, respectively. They predict that consumers will respond to the expected marginal price or even less precise information about marginal price (Borenstein 2009). In turn, Liebman and Zeckhauser (2004) introduce in their model the assumption that consumers are not perfect optimisers (since they do not fully understand the nature of block tariffs) and find that the price they perceive corresponds to the average price at their consumption level.

Our objective with this paper is to work with a price escalation metric that reflects the suitability of a tariff to promote water conservation. While reacting to marginal prices requires that consumers be perfectly informed about the price

[39] That is, the difference between the actual water bill paid by consumers and the amount they would have paid had all units consumed been charged at the same price as the marginal unit.

[40] For example, a heat wave that increases water needs or unexpected bonuses and dividends for income earners may prevent them from determining their income in advance and optimise consumption.

structure they face, reacting to average prices requires much less information. In the latter case, only information about the total bill and quantity consumed, readily accessible from any standard bill, is needed. Since in practice most consumers usually admit to not knowing their tariffs (Pérez-Urdiales *et al.* 2015), we choose to use average price as the price variable of reference with which to construct our proposed index of price escalation. Moreover, this theoretical prediction about the average price being the price perceived by consumers has been confirmed by numerous empirical (Shin 1985; Liebman and Zeckhauser 2004; Ito 2014) and experimental studies (De Bartolome 1995).

2.2.2. Two additional considerations.

Before further considering the analysis of price escalation, two more concerns must be addressed: the effects of a fixed component and the choice of the importance that should be attached to each of the price averages considered under the nonlinear tariffs.

2.2.2.1. Treatment of the tariff's fixed component

The adoption of two-part tariffs, including both a fixed fee and a variable or volumetric fee, is increasingly common (OECD 2010; GWI 2014; IWA 2014; Dinar *et al.* 2015). However, although necessary for the utilities' financial stability, the fixed component introduces a regressive element in the tariff. With a fixed fee, the effective average price per cubic metre will be relatively high and decreasing for the lowest levels of consumption, until the counterbalancing effect of the escalation embedded in the block prices makes it rise again. The overall degree of price escalation in the tariff effectively involves the combined effect of both tariff components, fixed and variable, so it is important to account for the fixed component in the proposed analysis.[41]

[41] The treatment of the fixed fee in the calculation of the price escalation index is one of the aspects of the current paper that deviates from the analysis in Suárez-Varela et al. (2015).

2.2.2.2. Considering proportions of users per block when dealing with IBRs.

When using aggregate data (usually collected at the municipality level), as it is the case in our study, water demand functions must be estimated by regressing average consumption against a measure of the price faced by the consumers in each jurisdiction. However, since most often water tariffs are non-linear, it is not obvious what the relevant price should be.[42] Under these conditions, using an average of the different marginal prices in the tariff weighted by the proportions of users consuming in each corresponding block has been acknowledged as the most theoretically correct specification for the analysis of the effects of IBRs on demand when only aggregate data is available (Schefter and David 1995; Martínez-Espiñeira 2003).[43]

As explained in Section 2.1, in our analysis the average price is more relevant. However, as our proposed measure of price escalation is intended as an aggregate indicator of the level of price escalation in water tariffs, we do follow a strategy analogous to the one described above, namely weighing the degree of escalation built into the different ranges of consumption by the proportion of users that consume within each particular range. This way, the degree of escalation embedded in each of the intervals is given an importance in accordance with its frequency of occurrence.

2.3. Measuring price escalation in water tariffs

The comparative analysis of the degree of price escalation in water tariffs requires the development of synthetic indices that provide a common metric to help those in charge of designing and assessing water pricing policies. However, to our knowledge, no such indices have been developed. The recent contribution by Suárez-Varela *et al.* (2015) is the only attempt in this regard. We build on their work, proposing a new variant of their suggested measure that incorporates in its calculation widely available additional relevant elements. For the purposes of this paper, we define price escalation as the increase in the average price per cubic metre

[42] See, for example, Martínez-Espiñeira (2002) and Arbués et al. (2003) for further details on this issue and other difficulties associated with the estimation of water demand functions.

[43] See Worthington and Hoffman (2008) for further details in the history of water demand specification.

as consumption increases. Hence, in order to measure price escalation, we must measure the average price that the tariff in use in each jurisdiction yields at different levels of consumption.

Water is consumed to satisfy different types of needs and wants, so it can be viewed differently from an equity perspective, all the way from being seen as a basic good to a luxury good, depending on the amount consumed. In order to account for the different uses of water (basic, intermediate, and luxury), we work out from each tariff the average price of water for a number of different levels of consumption and attach different weights to each, before incorporating them into our measure of price escalation. Following Martínez-Espiñeira *et al.* (2012), we consider the average variable price per cubic metre (excluding fixed costs) corresponding to a hypothetical bill of 3, 5, 10, 15, 20, 25, and 50 cubic metres (which we denote $bill_c$). Then, as formally explained below, we calculate the differences between the average prices in subsequent levels of consumption. We then weight these differences by the proportion of users within the range of consumption included between the two relevant consumption levels and finally we sum them up. By weighting the elements of the summation, our objective is that the resulting variable reflect the real effect of the price escalation embedded in the tariff in containing demand. For instance, a sharp escalation might be embedded in the price difference between two blocks, but if no consumers fell within that range, the impact of that price escalation on water conservation would indeed be effectively much weaker, if not null. The value yielded by the previous computations is also normalized employing the average price per cubic metre at 25 m^3 (that is, the middle point of the range considered). This normalization corrects the effect of a higher average level of prices in a municipality, which would systematically lead to higher values of the outcome variable, unduly distorting the comparison of price escalation across jurisdictions.

In summary, the suggested measure is obtained as follows:

$$ESC = \frac{\sum_{c=5}^{50} \left(\frac{bill_c}{c} - \frac{bill_{c-1}}{c-1}\right) \; w_{\frac{c}{c-1}}}{\frac{bill_{25}}{25}} \tag{13}$$

$$for\ c = 3m^3, 5m^3, 10m^3, 15m^3, 20m^3, 25m^3, 50m^3$$

$$and \quad w_{c/(c-1)}\ being\ the\ proportion\ of\ users\ consuming\ between\ c-1\ and\ c$$

Finally, a thorough analysis of the level of price escalation in the tariffs must also include the influence of the size of the fixed component. Therefore, a parallel measure of price escalation, analogous to the previous one, is proposed. In this case, average unit prices are computed including also the fixed component in the total estimated bill (*totbill*).

$$ESCfix = \frac{\sum_{c=5}^{50}\left(\frac{totbill_c}{c} - \frac{totbill_{c-1}}{c-1}\right) w_{\frac{c}{c-1}}}{\frac{totbill_{25}}{25}} \tag{14}$$

$$for\ c = 3m^3, 5m^3, 10m^3, 15m^3, 20m^3, 25m^3, 50m^3$$

$$and\ \ w_{c/(c-1)}\ being\ the\ proportion\ of\ users\ consuming\ between\ c-1\ and\ c$$

The previously depicted measures are intended to be complementary. As explained above, in a two-part tariff scheme, the variable component targets efficiency, sustainability, and equity goals. However, the fixed part is expected to ensure the financial stability of the utility, contributing to cost recovery goals. Given that the fixed component introduces a regressive element within the first levels of consumption, its inclusion attenuates the value of the measure.[44] Therefore, the escalation measure that includes only the variable component (*ESC*) is expected to reflect the efforts made to promote water conservation habits and to meet the objectives related to the variable component, while *ESCfix* illustrates the effective overall degree of price escalation embedded in the tariff after the specific goals of this component[45] are accomplished.

Finally, it should be noted that the two measures proposed yield different values depending on the nature of the pricing scheme that underlies the tariff. In the case of tariffs based on decreasing block rates or flat rates, average price is decreasing in consumption. Therefore, differences between average prices at

[44] Recall that, in this case, differences between average prices in subsequent levels of consumption is negative until the counterbalancing effect of the escalation embedded in the variable part turns it into positive. Therefore, the negative differences may be compensated by the positive ones, attenuating the value of the measure.

[45] It must be noted that the cost recovery goal is paramount in many regulatory regimes (i.e. Water Framework Directive in Europe).

subsequent levels of consumption are negative, leading to negative values for both measures and indicating that those tariffs are in fact regressive on consumption. For volumetric proportional rates, a constant average price makes differences between subsequent ranges of consumption null, thus leading to a null value for the measure as well. And finally, those in IBRs present a positive value for the indexes and its size will increase with the degree of price escalation embedded in the tariff.

2.4. The analysis of price escalation in water tariffs. An empirical implementation for Spain.

Spain is a Southern European country subject to water stress[46] or severe water stress throughout most of its territory. In fact, it is the third country of the European Union with the highest water exploitation index[47] and the incidence of droughts is expected to increase in the near future (OECD 2011). Regarding domestic supply, Spain is also one of the countries with the highest per capita consumption (Eurostat), although significant efforts have been made to reduce it during the last decade (Instituto Nacional de Estadística 2014).

In Spain water prices are set at the municipal level, since each municipality has jurisdiction over the management of its water service. There does not exist a legal framework or regulatory body that establish common guidelines for the determination of water tariffs, so municipalities are left with the decision about how to design their own price schedules. Therefore, tariff systems are fairly diverse (González-Gómez *et al.* 2012). In the absence of a formal regulatory body, the Spanish Association of Water Supply and Wastewater (AEAS), which is financed by both private and public utilities and public institutions and is the Spanish representative member of the International Water Association (IWA) and the main advisory body in the Spanish water sector, elaborates the main statistics and indicators related to the performance of the water sector, as well as some recommendations for the design of water tariffs (e.g. AEAS 2014).

Regarding the regulatory framework, the Water Framework Directive (WFD henceforth) that came into effect in 2000 places special emphasis on the

[46] According to the European Environmental Agency (EEA 2012, p. 40), a location is "water stressed" if its Water Exploitation Index (WEI) – defined as the ratio of annual freshwater abstraction to long-term water availability- is higher than 20%.

[47] The ratio of total water abstraction per year to total long term renewable resources.

maintenance and improvement of the quality of water bodies and the sustainable use of water resources, which confers a marked ecological and environmental character to this regulation (Petersen *et al.* 2009). More specifically, the use of economic instruments is highlighted as the main tool for the attainment of these objectives (Unnerstall 2007). According to Article 9 of the WFD, pricing policies should be granted a more prominent role in national policies addressed at covering the cost of the service and making a more efficient and sustainable use of the resource. In response, a volumetric tariff with a certain degree of escalation is usually recommended, justified mainly because it helps promote economic efficiency and sustainability goals (European Commission 2000; AEAS 2014). Nevertheless, according to the European Environmental Agency the efforts made by Member States in terms of pricing policy reform have been limited in practice to meeting the requirement of recovering supply costs, but "generally speaking, the WFD did not result in a change in water pricing policy" (European Environmental Agency 2013, p.9). Only in recent years have some countries, such as the Netherlands and Spain, implemented some changes in response to the WFD (European Environmental Agency 2013).

In Spain, according to AEAS, between 2002 and 2012 water supply tariffs experienced an increase of 7% (AEAS 2013).[48] Nevertheless, there is no information available about the changes made to the structure of the tariffs and its potential contribution to moderating consumption. In this context, we wonder whether the necessary changes in the structure of water pricing schemes have followed the rise in average water prices and whether price escalation has increased.

We will use the synthetic indexes proposed in Section 2.3. to evaluate the degree of price escalation in water supply tariffs in the years 2000 (when the WFD came into effect) and 2014, as well as the main factors affecting changes in price escalation during that period. That will allow us to assess the degree of implementation of the guidelines proposed by the WFD and identify the common characteristics of those municipalities that embedded a higher degree of escalation into their water tariffs.

[48] This weak increase as compared to other European countries may explain the abovementioned paradox that, despite being one of the countries of the European Union with a higher water exploitation index, Spain is also one of the countries with higher per capita consumption.

2.4.1. Data and Sample

This study uses data from the 2000 and 2014 water tariffs applied in 952 Spanish municipalities. This sample includes 11.91% of Spain's municipalities and covers around half (48.58%) of its total population. Together with this information on water tariffs, information on other variables that affect the degree of price escalation within each tariff and its evolution was added to the database. Table 21 in the Appendix contains a comprehensive description of the variables and data sources. All the variables refer to 2000 and 2014, except for those about water stress, as they reflect averages for longer periods. The prices that refer to 2014 have been deflated to their 2000 equivalents.

We expected water stress to be a main driver of price escalation in water tariffs, so we gathered information on the percentage of occupancy of reservoirs in the relevant river basin (*Occup*), August temperatures (*Temp*) and annual rainfall per square metre (*Pluv*), all expressed in averages for the periods 1990-1999 and 2010-2013. We also include a proxy for the level of economic activity (*Econ*) in the form of the index of municipal economic activity calculated by La Caixa (2014). Likewise, tourist activity was deemed to be a relevant factor for two reasons. First, because it increases the seasonality of demand, so to meet peak demands it is necessary to maintain excess capacity at other times. Second, because tourism in Spain is highly linked to water use, for example to fill pools or water golf courses. To proxy for it, we use the Tourist Activity Index (*Tourism*), also calculated in the 2014 edition of *La Caixa*'s Yearbook (La Caixa 2014).

The literature on public local services also points out that ideological and political factors can interfere with the management of those services, so they must be considered (Picazo-Tadeo *et al.* 2012). Left-wing parties are expected to be more committed to social and environmental causes, and therefore more likely to adopt a higher degree of escalation in their tariffs. In Spain, the centre-right is occupied by *Partido Popular* (*PP*), while the *Partido Socialista Obrero Español* (*PSOE*) represents the centre-left, and *Izquierda Unida* (*IU*) is a party located further to the left than PSOE in the political spectrum. Moreover, these national parties coexist with some other smaller parties with a more regional character. To test for the influence of ideological factors, we include binary indicators for the main parties in Spain (variables *PP*, *PSOE* and *IU*), taking value 1 if the party dominated the local government during the last three four-year ruling periods.[49] Also included as

[49] That implies that the party has ruled with majority for at least 12 out of the 14 years period considered in our sample.

factors related to political determinants, we include the degree of public support garnered by the ruling party (variable *Majority*) in each municipality is considered. This binary indicator equals 1 if during the previous three ruling periods the most voted party obtained over 50% of votes.

Additionally,[50] the type of ownership of the water service supplier may be a cause of heterogeneity among local utilities and, more precisely, in the water supply sector (Renzetti and Dupont 2009). Spanish law stipulates that the municipality is responsible for the management of water supply services but the local government has the authority to outsource the management to private companies and public-private enterprises. Therefore, we included a binary indicator of the type of ownership of the water utility serving the municipality (*Private*), with value of 1 if the utility is privately managed.

We also include several socioeconomic characteristics that serve both as controls and to test several of our hypotheses. These include indicators of population (*Population*), population density (*Density*), average household size (*HouseholdSize*) and unemployment rates (*Unemp*). Finally, and in order to account for the rest of components of the tariff, we gathered information on the number of blocks (*Blocks*) and the size of the fixed component of the tariff (*Fixed*).

Table 15 contains selected descriptive statistics.

Table 15: Summary statistics.

VARIABLES	N	Mean	St. Dev.	Min	Max
*Fixed*_2014	1,001	3.659	2.161	0	17.50
*Fixed*_2000	1,001	3.227	2.206	0	12.82
*Blocks*_2014	1,001	2.954	1.239	1	7
*Blocks*_2000	1,000	2.801	1.251	0	15

[50] Other factors, such as the percentage of water losses and the presence of economies of scale and scope are sometimes considered as factors influencing water prices (Carvalho and Marques 2013). However, in our study, the influence of the price level itself is offset by our normalizing of the measure of price escalation (as explained in Section 2.3.), so that the municipalities with a higher average level of price do not unduly yield systematically higher values of the proposed escalated measures.

*Occup*_2000_2013	1,001	69.56	16.92	0	90.36
*Occup*_1990_1999	1,001	57.13	18.32	0	83.09
*Pluv*_2000_2013	1,001	639.6	248.5	279.9	1,512
*Pluv*_1990_1999	1,001	627.0	251.6	265.1	1,418
*Temp*_2000_2013	988	22.87	2.468	17.94	26.39
*Temp*_1990_1999	1,001	22.67	2.134	17.88	26.12
*Density*_2014	994	0.629	1.873	0.001	24.55
*Density*_2000	994	0.513	1.591	0.001	17.75
*Population*_2014	1,001	25.39	127.0	0	3,234
*Population*_2000	1,001	21.91	114.8	0.021	2,883
Tourism_2014	1,000	46.14	407.4	0	9,400
*Tourism*_2000	992	39.44	372.4	0	9,497
Unemp_2014	1,000	10.80	7.372	0	28.30
*Unemp*_2000	1,000	3.743	2.876	0	14.80
*Econ*_2013	1,000	58.176	414.45	0	10,488
*Econ*_2000	992	55.77	388.0	0	9,639
Householdsize_2014	961	2.515	0.276	1.493	3.769
*Householdsize*_2000	965	2.808	0.296	1.533	3.769
*Private*_2014	1,001	0.239	0.427	0	1
*Private*_2000	1,001	0.262	0.440	0	1
PP_2000_2014	994	0.202	0.402	0	1
PP_1988_2000	1,001	0.248	0.432	0	1
PSOE_2000_2014	994	0.0996	0.300	0	1
PSOE_1988_2000	1,001	0.383	0.486	0	1
IU_2000_2014	993	0.00806	0.0894	0	1
IU_1988_2000	1,001	0.0340	0.181	0	1
*Majority*_2000_2014	994	0.347	0.476	0	1
*Majority*_1988_2000	991	0.291	0.454	0	1

2.4.2. The impact on the WFD on water tariffs and price escalation.

In this section, the two proposed indexes are used to conduct an analysis of the degree of price escalation in Spanish water tariffs in 2000 and 2014, as well as to assess the evolution showed by these indicators between those two reference points.

First, we calculated the total variable cost and the total bill (that is including the fixed component of the tariff) corresponding to the hypothetical monthly bills for consumption levels of 3, 5, 10, 15, 20, 25, and 50 cubic metres.[51] Table 16 shows the main descriptive statistics related to these variables, as well as *Fixed* and *Blocks* for the two years considered, together with the percentage change in their mean values between 2000 and 2014.

Table 16: Descriptive statistics of variables Fixed, Blocks, $Bill_c$ and $Totbill_c$.

Variable	Year	Mean	Std. Dev.	Min	Max	% Increase in mean values
Fixed	2000	3.27	2.23	0	12.82	13.76%
	2014	3.72	2.18	0	17.50	
Blocks	2000	2.82	1.24	0	15.00	4.96%
	2014	2.96	1.22	1	7	
Bill3	2000	1.02	1.00	0	6.41	6.86%
	2014	1.09	0.79	0	8.28	
Bill5	2000	1.81	1.85	0	19.99	2.21%
	2014	1.85	1.25	0	7.50	
Bill10	2000	4.25	3.73	0	33.13	2.82%
	2014	4.37	2.68	0	20.00	
Bill15	2000	7.30	5.76	0	54.64	8.49%
	2014	7.92	4.53	0	32.50	

[51] We refer to section 2.3. for further information on the choice of these particular amounts.

Bill20	2000	10.99	7.99	0	76.15	12.10%
	2014	12.32	7.20	0	45.00	
Bill25	2000	15.29	10.93	0	97.66	14.72%
	2014	17.54	10.63	0.95	60.90	
Bill50	2000	40.77	29.55	0	243.49	20.55%
	2014	49.15	30.51	4.50	180.14	
Totbill3	2000	4.30	2.56	0.34	14.33	11.86%
	2014	4.81	2.35	0.56	17.92	
Totbill5	2000	5.08	3.04	0.34	21.43	9.65%
	2014	5.57	2.58	0.74	18.20	
Totbill10	2000	7.52	4.50	0.72	33.99	7.58%
	2014	8.09	3.54	1.05	26.26	
Totbill15	2000	10.57	6.34	0.72	55.50	10.12%
	2014	11.64	5.23	1.64	38.76	
Totbill20	2000	14.26	8.50	0.72	77.01	12.48%
	2014	16.04	7.81	2.09	51.26	
Totbill25	2000	18.56	11.35	0.72	98.52	14.55%
	2014	21.26	11.15	2.54	64.94	
Totbill50	2000	44.04	29.98	0.72	246.08	20.03%
	2014	52.86	31.15	4.79	182.56	

Then, as explained in Section 2.3., average prices at each level of consumption were computed based on both the total variable cost (*Bill*) and the total bill (*Totbill*) before the differences between subsequent thresholds were summed, weighted by the proportion of consumers falling within each consumption range. These weights rely on information about the number of users consuming water within each range. Ideally, these proportions should be obtained at the municipal level. Municipal civil servants and statistical agencies interested in implementing these measures should request information to utilities so that they reflect real proportions in the municipality or management unit being analysed. Unfortunately, in Spain this information is rarely available to researchers and we could not obtain it. Therefore, to illustrate the use of the measure, we use a rough approximation.

In Spain, the *Encuesta de Presupuestos Familiares* (Consumer Budgets Survey)[52] conducted by the Spanish National Statistical Service (Instituto Nacional de Estadística 2006-2012) gathers information about the number of cubic metres consumed by each household. This can, therefore, be used to obtain the distribution of users per range of consumption. We can introduce added variability in these proportions, since households can be classified in 570 groups[53] or clusters according to the characteristics of their place of residence; including population density, range of total population, region (which one of Spanish Autonomous Communities), and whether the municipality is a capital of a province or not. That allows us to simulate the proportions of consumers per consumption range and impute them to the municipalities in our database, based on those aforementioned characteristics that are included in both databases. The proportions of users within each range of consumption in our sample are shown in Table 17. Finally, the normalization explained in Section 2.3. is applied to both *ESC* and *ESCfix.* Table 18 shows descriptive statistics for the two indicators of price escalation in the two considered periods. The absolute differences in the indicators are also shown in the table (*Diffprog* and *Diffprogfix* respectively).

Table 17: Proportions of users within each range of consumption.

Variable	Prop 0-3	Prop 3-5	Prop 5-10	Prop 10-15	Prop 15-20	Prop 20-25	Prop 25-50	Prop50+
Value	11.99%	13.86%	37.04%	19.22%	8.61%	3.82%	4.64%	0.82%

[52] Consumer budget surveys are developed in many countries and they are intended to constitute a representative sample of the households within the scope of the survey. Therefore, these surveys could constitute a good source of information for our analysis as they can provide a rough approximation in cases in which information on proportions of users is not easily available.

[53] In our calculations, we rely on data from around 151,000 households. It should be noted that the Spanish Consumer Budget Surveys gather information about the number of cubic metres consumed by each household. In these surveys, the identifier of the municipality is masked in order to guarantee the anonymity of the households, so we could not estimate the proportion of consumers falling within each block on a municipal basis using a sample.

Table 18: Evolution of price escalation indices between 2000 and 2014.

Variable	Year	Mean	Std. Dev.	Min	Max
ESC	2000	0.08686	0.08618	-0.16238	0.89383
	2014	0.08656	0.08844	-0.76116	0.55577
ESC*fix*	2000	-0.24941	0.23050	-1.37185	0.58351
	2014	-0.24537	0.20324	-1.50048	0.42608
Diffprog		-0.0039	0.09277	-0.88849	0.45054
Diffprogfix		0.0404	0.20528	-1.42663	1.19306

With respect to the impact of the WFD, Table 16 allows us to analyse the impact on the water tariffs themselves. Our figures show that, after the implementation of the WFD and following the guidelines of the main advisory bodies,[54] the municipalities have significantly increased the fixed component, as well as the number of blocks (13.76% and 4.96% respectively) in their tariffs. We also find that the variable price of water has increased, on average, for every consumption level considered in our study. Moreover, the consumption levels that have experienced a greater price increase are those above 20 m^3, and the higher the consumption level, the higher the percentage increase, being 50 m^3 the one that has experienced the sharpest price increase. It should also be noted that, contrary to expectations, the prices at 3 m^3 have also increased substantially, as opposed to subsequent consumption levels. After the implementation of the WFD and under the recommendation of the main advisory body in Spain,[55] many municipalities decided to eliminate free allowances, as they were regarded as counterproductive in promoting an efficient and sustainable consumption of the resource. In our sample, the number of municipalities including these free allowances actually decreased from 196 in 2000 to 139 in 2014. Thus, our results seem to suggest that local governments have made a substantial effort after the enforcement of the WFD to increase price escalation by increasing prices more sharply in the higher levels of consumption, while some regressivity was introduced as a consequence of the elimination of the free allowances.

[54] See AEAS (2014) for these and other recommendations.

[55] See, for example AEAS (2014).

Nevertheless, despite the formal efforts made to increase price escalation in the tariff schedules, as shown by Table 17, the main increments in water tariffs took place in those ranges of consumption with lower proportions of consumers. Thus, when the actual distribution of consumption is considered, the picture we obtain is rather different.

Our proposed measures provide a synthetic index of the degree of price escalation with which to study the level of price escalation in each period, as well as the evolution between different periods. The figures in Table 18 lead us to highlight two facts. First, if we only take into account the variable component and once the distribution of consumers is accounted for,[56] tariffs have on average a positive degree of escalation embedded, both before and after the implementation of the WFD (the values of *ESC* are positive). However, if we include the effect of the fixed component, tariffs are regressive in consumption on average (as revealed by the negative mean value of *ESCfix*). Interestingly, we can also see that, accounting for both the fixed and variable components, some progress has been made to increase price escalation (the differences in *ESCfix* are positive on average), while there has actually been a decrease in the degree of price escalation in the variable component of the tariffs between 2000 and 2014. This would be the opposite of what would have been expected from the implementation of the WFD.

2.4.3. Determinants of the evolution of price escalation in water tariffs.

The second purpose of our empirical analysis is twofold. First, we analyse the determinants of the evolution of price escalation in water tariffs from 2000 to 2014. Second, we try to identify the characteristics of the municipalities that have more successfully increased, in line with the guidelines set by the WFD, the degree of price escalation. We, therefore, focus on the changes in price escalation levels brought about the coming to effect of the WFD.

Moreover, several of the variables introduced in Section 2.4.1. as relevant explanatory factors of price escalation are expressed in terms of differences

[56] Recall that in our proposed measure of price escalation the differences between the average prices in subsequent levels of consumption were weighted by the proportion of users within that range of consumption in order to reflect the real effect of the price escalation embedded in the tariff in containing demand.

between the two periods considered (before and after the implementation of the WFD). That is the case of DFixed DBlocks, DOccup, DPluv, DTemp, DDensity, DPopulation, DTourism, DUnemp, DEcon, DHsize, Privthroughout, Continuity and *Table 22* and *Table 23* include further details about the definition and summary statistics of these variables.

2.4.3.1. Methodology

After calculating the degrees of price escalation before (that is, in 2000) and after (2014) the water pricing policy reforms introduced by the Water Framework Directive (WFD), a variable measuring the differences for each municipality between those two time periods was constructed.

A difference between two continuous variables, the resulting variables was itself continuous too, so the simplest way to analyse it would be to use an ordinary least squares (OLS) estimator. However, a simplified version of the original dependent variable (*DiffESC*) was constructed by turning it into a trichotomous categorical variable (*policychoice*) that reflects the three possible general choices available to the municipalities. That is, each jurisdiction could have increased the level of price escalation, maintained it at the same level or even decreased it during the period considered. This variable was, therefore, obtained at the expense of sacrificing some of the information contained in the data but allowed us to uncover more significant relationships between the explanatory factors and the three resulting ordinal values of *policychoice* (indicators of a decrease, no change, and an increase in price escalation). Table 19 shows the frequency distribution of this variable.

Table 19: Frequency distribution of variable policychoice.

	Absolute frequency	Percent	Cum.
"Decreasing"	312	31.17	31.17
"Unchanged"	262	26.17	57.34
"Increasing "	427	42.66	100.00
TOTAL	1,001	100.00	

We note that *DiffESC* was simplified into *policychoice* after substituting by zero those values of the *DiffESC* that were smaller than 0.001 in absolute value. Otherwise, only one municipality would have the value zero, suggesting, spuriously, that only that municipality had experienced no changes in price escalation between 2000 and 2014. This adjustment was needed because, during the considered period, a key change occurred in the European Union. From 2002, the euro became the official currency in several European countries. Therefore, when we were constructing the price variable, we had to perform a currency conversion to homogenize the water bill amounts from 2000's pesetas (the former Spanish currency) to euros. To this end, we used the official exchange rate set by the monetary authorities at the beginning of the euro era: 166.386 pesetas/euro. However, many municipalities, when revising their tariffs, simplified instead the resulting amounts by rounding the result of the conversion to either the second or third place after the decimal point. By applying the correction mentioned above, we make sure that we offset the spurious effect of that rounding in those municipalities that have not suffered an actual increase in tariffs during the period.

The immediately obvious way to model variable *policychoice* would be to use an ordered logit or ordered probit estimator. Ordered regression models rely on the assumption that, although the effect of a given explanatory variable on the probability of any given outcome is not constant (depending instead on the values of all explanatory variables), there is an overall direction of the effect of the explanatory variables and proportionality in the odds of choosing among different categories. This parallel regression assumption,[57] more specifically, implies that the relationship between each pair of outcome groups is the same.[58] In other words, the ordered regression models assume that the coefficients that explain the choice between decreasing price escalation versus not changing it are the same as those that explain the choice between the latter and the choice to increase the degree of price escalation. Under these conditions, only one set of coefficients needs to be estimated to explain the probabilities of all outcomes. Otherwise, two different models would be needed to describe the relationship between the three possible outcomes.

The assumption makes it possible to exploit a very elegant and parsimonious model from which a generic interpretation of coefficients is straightforward. However, the assumption is often violated in practice. In fact, a Likelihood-Ratio test of proportionality of odds across response categories showed that the parallel

[57] See Long and Freese (2006, pp. 197-200) for further details.

[58] Expressed in terms of odds of choosing each of the categories, one can say that the proportional odds between two different categories are constant, independent of the categories considered.

regression assumption could be rejected at a 1% level, so less restrictive (and thus less parsimonious) models had to be considered. Generalized ordered models, such as the generalized ordered logit model obtained by Stata's *gologit* routine (Kang Fu 1997) or the "partial proportional odds" model (Peterson and Harrell 1990) are possible options but they can lead to interpretation issues and, if the notion of ordinality in the explained variable is preserved, problems of nonsensical predicted probabilities.[59,60] Moreover, generalized ordered logit estimations of our model did not show convergence to a unique solution.

In light of these difficulties, we chose to use a multinomial logit model, entirely giving up on the ordinal nature of the information and instead considering the three categories in the dependent variable as nominal outcomes. The multinomial logit model relies on the assumption of independence of irrelevant alternatives, which we also tested. [61]

The multinomial logit model allowed us also to conduct a test of the hypothesis that any two categories out of the three available (decreased price escalation, kept it unchanged, and increased it) could be combined. Both these tests suggested that it would be more efficient in our case to merge the categories indicating decreased and unchanged price escalation and model the dependent variable in its simplest form, namely as a binary variable. We, therefore, used a logit model to explain whether the municipalities' decision to increase price escalation (rather than decreasing it or leaving it unchanged).

2.5. Results

Table 20 displays the average marginal effect for all the variables included in the Logit model whose dependent variable is an indicator of whether the municipality increased price escalation instead of decreasing it or keeping it at the same level between 2000 and 2014. Unlike linear regression models, the coefficients from non-linear outcome models (such as logit) are not directly interpretable as

[60] This is because the regression lines no longer forced to run parallel must cross at some point. When that issue affects a nontrivial number of cases in within-sample values of the explanatory variables, using a generalized ordered model is problematic (McCullagh and Nelder 1989, p. 155).

[61] The Hausman test did not reject the null hypothesis of IIA. For some of the categories, $\chi 2$ values of the test statistic displayed negative values. This is very common and has been found as evidence against violation of the IIA assumption (Hausman and McFadden 1984).

marginal effects. Traditionally, marginal effects are reported at the mean values of the variables included in the model, that is, for the "average individual" in the sample. Although computationally more demanding, the interpretation of average marginal effects (the average of partial changes over all observations) is preferred by many authors (Bartus 2005). With the development of new statistical packages and increasing computational capacity, average marginal effects have become increasingly used and the most common form to report and interpret discrete choice models. It is this second type of marginal effects that we report in Table 20. A Likelihood-Ratio test of the hypothesis that all coefficients except the intercept are zero is rejected at 1%, implying that the model is globally significant.

Table 20: Average marginal effects from binary logit.

	Average Marginal Effects
DEcon	-0.000761
Econ_2014	0.000336
DHsize	-0.0667
Houseoldsize_2014	-0.108*
DPopulation	0.0127***
Population	0.000617
DDensity	-0.00647
Density	-0.0152
DOccup	-0.0130***
DPluv	-0.000189
DTemp	-0.00246
Occup	-0.000389
Pluv	0.000230*
Temp	0.0896***
Privatethroughout	-0.0135
Continuity	-0.387***
Dpubpriv	-0.481***

Tourism_2014	-0.000837**
DTourism	-0.000522
DUnemp	0.0115
Unemp_2014	-0.00850
DBlocks	0.142***
DFixed	0.0295***
PSOE_2000_2014	-0.0520
PP_2000_2014	0.190**
IU_2000_2014	0.0900
Majority_2000___2014	-0.0826
Observations	952
Pseudo-R2	0.19
LR test	250.06***

Legend: * p<0.1; ** p<0.05; *** p<0.01

The results of the logit model show that those municipalities that have experienced a larger growth in population along the period are more likely to have increased the degree of price escalation. It is reasonable that those municipalities whose population has increased more also have stronger incentives to promote water demand contention, as a means to avoid network overload or the need for further investments to meet the increasing demand. Municipalities under more water stress would be expected to respond to an increased pressure on their current and future availability of resources by increasing price escalation to adjust their demand to an expected reduction in supply. Our results suggest that municipalities that face higher temperatures or a higher reduction in the level of reservoir occupancy during the studied period have increased the price escalation embedded in their tariffs. However, it seems that those municipalities with less abundant rain have not been so inclined to maintaining or decreasing the degree of price escalation instead.

Regarding political and ideological issues, those municipalities in which PP (the main right-wing party in Spain) has been ruling with a majority show a significantly higher probability of experiencing increases in price escalation. Although left wing parties might be expected to be more committed to social and environmental goals, and thus more prone to increasing the level of price escalation in the tariffs, it is not the first time that a result of this type is found. In fact, previous research has found that, although left-wing parties usually increase public expenditures, it is right-wing parties that tend to set more progressive taxes and benefits (Padovano and Turati 2012; Bucciol *et al.* 2013). Finally, those municipalities that have increased price escalation also tend to present an increase in the number of blocks and the fixed component of the tariff.

Conversely, we find that the impact of continuity in the type of ownership of the management during the period or a change from public to private management can decrease the probability of increasing price escalation in a 38.7% and 48.1% respectively. This is not surprising, as private managers should not be expected to be as committed to the environmental and equity benefits derived from an increased escalation in prices. Likewise, the continuity in the type of ownership provides companies with fewer incentives to improve management and efficiency in the use of water resources. Furthermore, a higher level of tourist activity is also related to lower probabilities of increasing price escalation. It should be noted that tourist activity in Spain is highly linked to the use of water, for filling pools or watering gardens and golf courses, for example. These uses for water could be regarded as a type of "luxury good" that would rely on the highest levels of consumption. Thus, it is to be expected that towns with a higher influx of tourists, local authorities try not to penalize this type of consumption by embedding a sharper level of escalation in the prices, as it could potentially harm the tourism industry. Finally, we find that municipalities with a higher average household size are less likely to have raised the degree of price escalation. This constitutes a positive fact, as otherwise price escalation would be penalizing larger families instead of promoting an efficient allocation of resources.

2.6. Conclusions

Water scarcity is expected to be one of the main challenges humanity will face in the coming years. Thus, the efficient and sustainable use of water resources is becoming increasingly relevant. Furthermore, water is a basic need, so universal access to at least a certain amount of water must be guaranteed at affordable prices. In the last decades, the use of economic tools has been fostered as the main tool for the achievement of these goals. In this line, a volumetric tariff with some degree of price escalation is usually recommended. The more the degree of price escalation in the tariff the more substantial its expected contribution to a better allocation of resources.

However, despite the widespread use of price escalation in water tariffs, no established procedure exists to measure its degree. In this paper, we propose a measure of the level of price escalation in water tariffs and demonstrate its usefulness by analysing the evolution of price escalation between 2000 and 2014 in residential water tariffs using a sample of 952 Spanish municipalities. We examine the factors influencing this evolution in the context of the guidelines included in the European Water Framework Directive (WFD), the main legislative body of reference in Spain.

We find that, when only the variable component is considered, the tariffs exhibit, on average, a positive degree of price escalation both before and after the implementation of the WFD. However, when the fixed component is also considered in the analysis, tariffs are found to be regressive, on average, in both periods. Moreover, our results suggest that, despite the formal efforts made by local governments to increase price escalation after the coming into effect of the WFD by increasing prices more sharply at higher levels of consumption, the main increments actually affected those ranges of consumption with smaller proportions of consumers. Thus, once the distribution of consumers is considered through the use of our proposed measures of price escalation, our results show that, accounting for both the fixed and variable components, some progress has been made in terms of price escalation, while there has actually been a decrease in the degree of price escalation in the variable component of the tariffs between 2000 and 2014, contrary to what should have been expected from the implementation of the WFD. Besides, we find that some factors related to water stress, ideological factors, socioeconomic characteristics, issues related to tourist activity, and the ownership of the management may be affecting the probability of adopting tariffs with a higher level of escalation.

The main policy recommendations that can be drawn from our results involves the need to improve the incorporation into national law and implementation of the WFD in Spain. As pointed out by Pinto and Marques (2015), the decentralization of price-setting decisions to the municipalities may be hindering the implementation of the WFD, as it leaves the determination of price structures to retail utilities that may lack the necessary technical capability to design tariffs that simultaneously fulfil the multiple objectives of cost recovery, efficiency, sustainability, and equity expected from water tariffs. Moreover, the decentralization of the design of water tariffs may lead to political interference in those price-setting strategies (Pinto and Marques 2015). Therefore, the creation of a regulatory body with the empowerment to standardize the design of pricing policies as well as stablishing clear guidelines is highly encouraged. The reader should refer, for example, to the cases of the OFWAT in UK (Ofwat 2013), the ERSAR in Portugal (Martins et al. 2013b) or the CER in Ireland (CER, 2014) for case studies in this respect.

In summary, our contribution with this paper is twofold. First, we suggest two synthetic indexes of the degree of the price escalation within water tariffs. Our intention is that these summary metrics assist policy makers in conducting sound policy analysis, when assessing efforts made by countries and/or municipalities to enhance the price escalation of tariffs or performing comparative analysis between jurisdictions. Additionally, our empirical application should help public administrators identify favourable scenarios for the implementation of the due improvements in pricing policies, thus directing their efforts towards generating those propitious conditions. Similar analyses could be conducted in other regions with slight changes to adapt it to the specific problems and the legal framework in those regions.

Appendix

Table 21: Variables: description and sources.

	Variable	Description	Source
DEPENDENT VARIABLES	*ESCESC*	See Section 2.3.	Own construction
	ESC*fix*ESC*fix*	See Section 2.3.	Own construction
WATER TARIFF	*Fixed*	Amount of the fixed fee of the tariff	Official Gazettes of the Provinces and Regions
	Blocks	Number of blocks in the variable part of the tariff	Official Gazettes of the Provinces and Regions
PROXIES FOR LEVELS OF WATER STRESS	*Occup*	Proportion of occupancy of reservoirs in the municipality's river basin. Average for 2000-13 (year 2014) and average for 1990-99 (year 2000)	Ministry of Agriculture, Food and Environment. Area of Hydrological Information
	Pluv	Annual rainfall per square metre. Average for 2000-13 (year 2014) and average for 1990-99 (year 2000)	Spanish National Meteorology Agency
	Temp	Average of August temperatures. Average for 2000-13 (for year 2014) and average for 1990-99 (year 2000)	Spanish National Meteorology Agency
SOCIO*ECON*OMIC CHARACTERISCS OF MUNICIPALITY	*Density*	Inhabitants per km^2 (in 1,000s)	National Institute of Statistics (INE) and National Geographic Institute
	Population	Number of inhabitants (in 1,000s)	Municipal Census and Statistical

			Yearbook of Spain. National
	Tourism	Tourist activity index	La Caixa (2014)
	Unemp	Unemployment rate (proportion)	Labour Force Survey. National Institute of Statistics (INE).
	Econ	Economic activity index	La Caixa (2014)
	Householdsize	Average household size (members per household)	National Institute of Statistics (INE)
SERVICE MANAGEMENT	*Private*	Binary variable= 1 if private management and 0 in case of public management	Water utilities
IDEOLOGY	PP	Binary variable = 1 if PP has dominated the local government during at least 2 of the last three four-year ruling periods and 0 otherwise	Home Office and Ministry of Regional Policy
	PSOE	Binary variable = 1 if PSOE has dominated the local government during at least two of the last three four-year ruling periods and 0 otherwise	Home Office and Ministry of Regional Policy
	IU	Binary variable = 1 if IU has dominated the local government during at least two of the last three four-year ruling periods and 0 otherwise	Home Office and Ministry of Regional Policy
POLICY	*Majority*	Binary variable = 1 if during the last three ruling periods the most voted party secured over 50% of the votes and 0 otherwise	Home Office and Ministry of Regional Policy

All the variables included in this appendix were gathered with reference to both 2000 and 2014. The only exceptions are the variables related to weather (*Pluv, Temp, Occup*), which are means calculated over longer periods (see Table 21). It should also be noted that, following the description, in the case ideology and policy the variables related to 2014 reflect the situation during the last three four-year ruling periods (that is from 2002 to 2014), while the variable that makes reference to the year 2000 accounts for the three four-year ruling periods before 2000.

Table 22: Variables in differences for the two periods considered, description and sources.

	Variable	Description
Dependent variables	*DiffESCg*	Change in price escalation along the period (Difference between ESCin 2014 and *ESC* in 2000).
Water tariff variables	*DFixed*	Difference in the fixed component between 2014 and 2000
	DBlocks	Difference in the number of blocks between 2014 and 2000
Proxies for levels of water stress	*DOccup*	Change in occupation in reservoirs during the considered period (Difference between *Occup*_2014 and *Occup*_2000).
	DPluv	Change in average rainfall during the considered period (Difference between *Pluv*_2014 and *Pluv*_2000).
	DTemp	Change in average rainfall during the considered period (Difference between *Temp*_2014 and *Temp*_2000)
Socioeconomic characteristics of the municipality	*DDensity*	Change in density of population during the considered period (Difference between *Density*_2014 and *Density*_2000)
	DPopulation	Change in population during the considered period (Difference between *Population*_2014 and *Population*_2000)
	DTourism	Change in tourist activity during the considered period (Difference between *Tourism_2014* and *Tourism*_2000)
	DUnemp	Change in unemployment rates during the considered period (Difference between *Tourism_2014* and *Tourism*_2000)

	DEcon	Change in economic activity during the considered period (Difference between *Econ_2014* and *Econ_*2000)
	DHsize	Change in household size during the considered period (Difference between *Houseoldsize_2014* y *Houseoldsize _2000*)
Service management	*Continuity*	Continuity in the type of ownership of the management along the period. The same manager has been managing the service along the period.
	Privthroughout	Management was private throughout the considered period (2000-2014).
	Dpubpriv	A change from public to private management occurred during the period (2000-2014)

Table 23: Summary statistics (variables in differences).

	N	Mean	St.Dev	Min	Max
DPopulation	1,001	3.485	13.76	-16.11	350.7
DHsize	961	-0.292	0.233	-1.105	0.780
DDensity	994	0.115	0.610	-1.798	17.55
DOccup	1,001	12.43	9.106	-2.359	36.74
DPluv	1,001	12.59	42.08	-91.07	143.6
DTemp	988	0.236	0.483	-0.560	0.984
DTourism	992	4.509	63.01	-455	1,615
DUnemp	1,000	7.058	5.159	-8.400	22
DBlocks	1,000	0.152	1.016	-11	5
DFixed	1,001	0.432	2.055	-10.91	16.42
DContinuity	1,001	0.825	0.380	0	1
Dpubpriv	1,001	0.152	0.359	0	1
Privthroughout	1,001	.2388	.4265	0	1

References

AEAS (2014). Guía de Tarifas de los Servicios de Abastecimiento y Saneamiento de Agua. Madrid, Spain.

AEAS (2013) Tarifas 2012. Precio de los Servicios de Abastecimiento y Saneamiento en España. Madrid, Spain.

Arbués, F., García-Valiñas, M.A. and Martínez-Espiñeira, R. (2003). Estimation of Residential Water Demand: A State of the Art Review. The Journal of Socio-Economics 32(1): 81-102.

Baerenklau, K. A., Schwabe, K., and Dinar, A. (2014). The residential water demand effect of increasing block rate water budgets. Land Economics, 90(4):683–699.

Bartus, T. (2005). Estimation of marginal effects using margeff. Stata Journal 5: 309-329.

Borenstein, S. (2009). To What Electricity Price Do Consumers Respond? Residential Demand Elasticity Under Increasing-Block Pricing. http://faculty.haas.berkeley.edu/borenste/download/NBER_SI_2009.pdf (accessed October 24, 2016).

Bovis, C.H. (2005). Financing services of general interest in the EU: How do public procurement and state aids interact to demarcate between market forces and protection?. European Law Journal 11(1): 79-109.

Bucciol, A., Cavalli, L., Pertile, P., Polin, V., and Sommacal, A. (2013). Redistribution at the Local Level: the Case of Public Childcare in Italy. Working Papers 21/2013. University of Verona, Department of Economics.

Carvalho, P., and Marques, R.C. (2013). Computing economies of vertical integration, economies of scope and economies of scale using partial frontier nonparametric methods. European Journal of Operational Research 234(1): 292-307.

Commission for Energy Regulation, CER (2014). Domestic Water Tariff Principles and Proposals. Consultation Paper CER/14/082.

De Bartolome, CAM (1995). Which Tax Rate Do People Use: Average or Marginal. Journal of Public Economics 56(1): 79–96.

Dinar, A., Pochat, V. and Albiac, J. (eds.) (2015). Water Pricing Experiences and Innovations. Springer Publications.

European Commission (2000). Pricing and sustainable management of water resources. Communication from the Commission to the Council, European Parliament and Economic and Social Committee [COM(2000) 477].

European Environmental Agency (2013). Assessment of cost recovery through water pricing. EEA Technical Report No 16/2013. Copenhagen, Denmark.

European Environmental Agency (2012). Towards efficient use of water resources in Europe. EEA Report No 1/2012. Copenhagen, Denmark.

Farolfi, S. and Gallego-Ayala, J. (2014). Domestic water access and pricing in urban areas of Mozambique: between equity and cost recovery for the provision of a vital resource. International Journal of Water Resources Development 30(4): 728-744.

Grafton, R.Q., Ward, M.B., To. H. and Kompas T (2011) Determinants of residential water consumption: Evidence and analysis from a 10-country household survey. Water Resources Research 47(8).

González-Gómez, F., García-Rubio, M.A. and Guardiola, J. (2012). Urban water service policies and management in Spain: Pending Issues. International Journal of Water Resources Development 28(1): 89–106.

GWI (2014). Global Water Market 2015: Meeting the world's water and wastewater needs until 2018. Global Water Intelligence, UK.

Hausman, J., and McFadden, D. (1984). Specification tests for the multinomial logit model. Econometrica 52 (5): 1219-1240.

Hewitt, J., and Hanemann, W. (1995). A discrete/continuous choice approach to residential water demand under block rate pricing. Land Economics, 71:173–192.

Instituto Nacional de Estadística (2006-2012) Encuesta de Presupuestos Familiares. Madrid. Spain

Ito, K. (2014). Do Consumers Respond to Marginal or Average Price? Evidence from Nonlinear Electricity Pricing. American Economic Review 104(2): 537–563

IWA (2014). International Statistics for Water Services. Statistics & Economics, International Water Association, London.

Kang Fu, V. (1997). GOLOGIT: Stata module to estimate generalized ordered logit models. Statistical Software Components, Boston College Department of Economics.

La Caixa (2014) Economic Yearbook of Spain. http://www.anuarieco.lacaixa.comunicacions.com/java/X?cgi=caixa.anuari99.util.ChangeLanguage&lang=eng

Liebman, J. B., and Zeckhauser, R. (2004). Schmeduling. www.hks.harvard.edu/jeffreyliebman/schmeduling.pdf (accessed April 03, 2017).

Long, J.S., and Freese, J. (2006). Regression Models for Categorical Dependent Variables Using Stata, Second Edition. Stata Press, College Station, TX.

Martínez-Espiñeira, R., García-Valiñas, M.A., and González-Gómez, F. (2012). Is the pricing of urban water services justifiably perceived as unequal among Spanish cities?. International Journal of Water Resources Development 28(1): 107-121.

Martínez-Espiñeira, R. (2003). Estimating Water Demand under Increasing-Block Tariffs Using Aggregate Data and Proportions of Users per Block. Environmental and Resource Economics 26(1): 5–23.

Martínez-Espiñeira, R. (2002). Residential water demand in the Northwest of Spain. Environmental and Resource Economics 21(2): 161-187.

Martins, R., Quintal, C., Cruz, L., and Barata, E. (2016). Water affordability issues in developed countries – The relevance of micro approaches. Utilities Policy 43(A): 17–123.

Martins, R., Cruz, L., Barata, E., Quintal, C. (2013a). Assessing social concerns in water tariffs. Water policy 15 (2): 193-211.

Martins, R., Cruz, L., and Barata, E. (2013b). Water Price Regulation: A Review of Portuguese Tariff Recommendations. Public Organization Review 13: 197-205.

McCullagh, P., and Nelder, J.A. (1989). Generalized Linear Models, Second Edition. Chapman and Hall/CRC.

Nataraj, S., and Hanneman, M. (2011). Does marginal price matter? A regression discontinuity approach to estimating water demand. Journal of Environmental Economics and Management 61(2): 198–212.

Nieswiadomy, M.L., Molina, D.J. (1989). Comparing residential water estimates under decreasing and increasing block rates using household data. Land Economics 65(3): 280–289.

Nordin, J.A. (1976). A proposed modification to Taylor's demand analysis comment. The Bell Journal of Economics 7(2): 719-721.

OECD (2011). Managing Water for All. An OECD Perspective on pricing and financing. OECD Publishing, Paris.

OECD (2010). Pricing water resources and water and sanitation services. OECD Publishing, Paris.

Ofwat (2013). Charging Principles and Guidelines. Planned Revisions Report. Ofwat, London.

Olmstead, S.M., Hanemann, W.M., and Stavins, R.N. (2007). Water demand under alternative price structures. Journal of Environmental Economics and Management 54(2): 181-198.

Pérez-Urdiales, M., García-Valiñas, M.A., and Martínez-Espiñeira, R. (2015). Responses to changes in domestic water tariff structures: a Latent Class Analysis on household-level data from Granada, Spain. Environmental and Resource Economics, in press.

Petersen, T., Klauer, B., and Manstetten, R. (2009). The environment as a challenge for governmental responsibility - The case of the European Water Framework Directive. Ecological Economics 68(7): 2058-2065.

Peterson, B., Harrell, F.E. (1990). Partial proportional odds models for ordinal response variables. Journal of the Royal Statistical Society. Series C (Applied Statistics) 39(2): 205-217.

Picazo-Tadeo, A.J., González-Gómez, F., Ruiz-Villaverde, A., and Guardiola Wanden-Berghe, J. (2012). Do ideological and political motives really matter in the public choice of local services management? Evidence from urban water services in Spain. Public Choice 151(1): 215-228.

Padovano, F., and Turati, G. (2012). Redistribution through a "leaky bucket". What explains the leakages?. In: Economics Working Paper from Condorcet Center for Political Economy at CREM-CNRS Condorcet Center for Political Economy.

Pinto, F. and Marques, R.C. (2015). Tariff recommendations: A panacea for the Portuguese Water Sector?. Utilities Policy 34: 36–44.

Renzetti, S., and Dupont, D. (2009). Measuring the Technical Efficiency of Municipal Water Suppliers: The Role of Environmental Factors. Land Economics 85(4): 627–636.

Rogers, P., Silva, R.D., and Bhatia, R. (2002) Water is an economic good: How to use prices to promote equity, efficiency, and sustainability. Water Policy 4(1): 1-17.

Saez, E. (2010). Do Taxpayers Bunch at Kink Points? American Economic Journal: Economic Policy 2(3): 180–212.

Saez, E. (1999). Do Taxpayers Bunch at Kink Points? National Bureau of Economic Research, Working Paper 7366.

Schefter, J.E., and David, E.L. (1985). Estimating residential water demand under multi-tariffs using aggregate data. Land Economics 61(3): 272–280.

Schoengold, K., and Zilberman, D. (2014). The economics of tiered pricing and cost functions: Are equity, cost recovery, and economic efficiency compatible goals?. Water Resources and Economics 7(1): 1-18.

Shin, J.S. (1985). Perception of price when information is costly: evidence from residential electricity demand. Review of Economics and Statistics 67(4): 591–598.

Suárez-Varela, M., Martínez-Espiñeira, R., and González-Gómez, F. (2015). An analysis of the price escalation of non-linear water tariffs for domestic uses in Spain. Utilities Policy 34(2015): 82-93.

Taylor, L.D. (1975). The Demand for Electricity: A Survey. The Bell Journal of Economics 6 (Spring): 74-110.

UN-Water (2006). Coping with water scarcity. Challenge of the Twenty-First Century. UN-Water, FAO, 2007.

Unnerstall, H. (2007). The principle of full cost recovery in the EU-water framework directive—genesis and content. Journal of Environmental Law 19(1): 29-42.

Van de Walle, S. (2008). What services are public? What aspects of performance are to be ranked? The case of “services of general interest”. International Public Management Journal 11(3): 256-274.

Wichelns, D. (2013). Enhancing the performance of water prices and tariff structures in achieving socially desirable outcomes. International Journal of Water Resources Development 29(3): 310-326.

World Economic Forum (2016). Global Risks 2016, 11th Edition. World Economic Forum, Geneva

Essay 3:
The Role of Environmental Attitudes on Averting Behaviors that Entail Negative Externalities: A Double-hurdle Approach Applied to Bottled Water Demand.

Abstract

In this paper, we explore the effect of individual's environmental attitudes on adopting mitigation actions that entail environmental negative externalities. Moreover, a double-hurdle approach is proposed to deal with the substantial number of zero consumption records in databases on averting behaviors. Using a dataset on bottled water consumption from two cities in southern Spain, we also explore the impact of several public policies related to residential water management that are being increasingly applied in many urban centers around the world. Our results suggest that some pricing and non-pricing water conservation

policies could result in environmentally undesirable effects derived from an increase in bottled water demand. We also find that fostering pro-environmental habits could prove very successful in curbing averting behaviors that pose environmental externalities. Additionally, we find that failing to properly address problems stemming from the large percentage of zero consumption records when modelling averting behaviors could give rise to misleading conclusions.

3.1. Introduction

Averting behaviors have been a long-standing matter of study. The existence of consumer's defensive responses to potential environmental or health impacts of pollution and other hazards has been long acknowledged. The term "averting behavior", also usually referred to as defensive or mitigating behavior, contains a wide range of actions with the common feature of being undertaken with the objective of either preventing exposure to certain environmental risks or hazards, or mitigating and compensating for their effects after exposure (Dickie, 2017). Some preventive actions would include, for example, the use of home air cleaners or purifiers for air pollution (Dickie and Gerking, 1991), using sunscreen lotion in order to reduce the risk of skin cancer (Murdoch and Thayer 1990) or installing water filtration systems and purchasing bottled water in order to avoid water contamination (Harrington et al. 1989; Zivin, 2011).

At an aggregate scale, the use of mitigating actions implies substantial costs for individuals and societies, which may involve monetary expenses, such as expenses on medical care for illnesses caused by air pollution (Gerking and Stanley, 1986) or purchase and installation of certain devices (Abdalla et al. 1992); time costs, as they usually imply change in daily activities (Neidell, 2009); as well as facing certain deprivations, such as reductions in outdoor time to avoid ozone exposure (Mansfield, Johnson and van Houtven, 2006). Some of these averting behaviors also entail environmental negative externalities, leading to great environmental costs. This may include generating waste and residuals that are for the most part non-biodegradable, such as plastic bottles and active carbon filters for water or masks for air pollution; energy needs associated with transport and for the use of certain devices (Deschenes and Greenstone, 2011), heating and air conditioning (Isaac and Van Vuuren, 2009) or toxic substances released to the atmosphere and maritime ecosystems severely affecting their sustainability (Farbaim et al. 2016; US EPA, 1998; Tovar-Sánchez et al. 2013). Particularly in relation to the empirical implementation of this paper, the bottled water industry is predicted to generate numerous environmental externalities. The amount of water needed to obtain one liter of bottled water amounts to 1.32 liters, contributing to the depletion of aquifers and spring waters (International Bottled Water Association, 2015). In addition, most plastic bottles are discharged after use into landfills (Arnold and Larsen, 2006). And energy needs associated with bottling and transport of the water significantly add to the environmental footprint, with a range of 5.6 to 10.2 Megajoules per liter (MJ/l) of bottled water.

The processes governing the decisions to undertake averting behaviors are complex and influenced by multiple objective and subjective factors that have been extensively studied. In this paper, we want to explore the role of environmental attitudes and behaviors. Given that, as mentioned earlier, some averting behaviors entail environmental negative externalities, it is to be expected that their choice over other more environmentally friendly alternatives would be influenced by individual's attitudes towards the environment.

Existing research claims that there exists a substantial gap between people's attitudes towards the environment and their actual actions (Blake, 1999). Thus, we study separately the influence of environmental attitudes and behaviors. In addition, with respect to environmental behaviors the analysis is further disaggregated in order to account for the different levels of individual pro-environmental involvement. Particularly, the two distinct classes of environmental behaviors the literature has usually identified as entailing different levels of sacrifice on the part of the individual are considered independently. Those include efficiency or one-shot behaviors such as installation of certain technologies addressed at saving resources, and curtailment behaviors including daily habits or sacrifices with the objective of preserving the environment (Stern and Gardner, 1981). This is important in terms of public policy, as policies aimed at fostering environmental concerns usually differ from the ones promoting behavioral change (Stern and Gardner, 1981). And the same applies to interventions tackling promotion of efficiency and curtailment actions.

One important issue when dealing with averting expenditures is the substantial percentage of households that do not consume any amount. Zero consumption may arise for several reasons and econometric modelling strategies will vary according to the economic interpretation placed on those zeros. However, this fact has been usually neglected in the literature. In this paper, we propose an empirical strategy to deal with the existence of a substantial number of zero consumption records in databases on averting behavior consumption and expenditures. Particularly, we use a generic double hurdle approach that allows us to model averting behaviours without departing from any particular hypothesis regarding the reasons why households do not adopt said behaviors –i.e. non-participation vs. corner solutions- and to test the underlying distributional assumptions in order to choose among specifications.

To meet our objectives, we use data on bottled water consumption from a 2014 household survey conducted in the towns of Baza and Guadix, in the province of Granada, Spain. Since bottled water is an averting behavior that poses a number of significant environmental negative externalities and can be substituted by other more environmentally friendly alternatives (e.g. filtering water), it can be expected that the decision to consume would be influenced by individual's attitudes and behaviors towards the environment. We observe that a significant number of household do not use bottled water as an averting behavior, thus posing the abovementioned feature of a substantial proportion of zero consumption records. The fact that Spain is the fourth largest consumer of bottled water per capita in Europe and the ninth in the world (Beverage Marketing Company, 2013) also makes it an interesting setting for our study. In addition, Baza and Guadix are located in the Guadalquivir River basin, which is under extreme water stress according to the European Environmental Agency (2012). Thus, efficient and sustainable management of water resources gets special attention from policy-makers in this region, and numerous water conservation policies are being applied to residential water supply. Given that bottled water could be a substitute to drinking from the tap, we analyze whether several pricing and non-pricing policies that are currently being applied in many urban centers around the world could be affecting bottled water demand, so that the potential environmental effects are accounted for when considering among the different types of public policies for residential water conservation.

The remainder of the paper is structured as follows. Section 3.2 presents an overview of the state of the art on the determinants of the adoption of averting behaviors and bottled water consumption. Our model is then outlined in Section 3.3. The data and methodological approach proposed for the empirical analysis are described in Section 3.4. Section 3.5 presents the results and the pertaining robustness checks, and Section 3.6 concludes with a discussion and policy implications.

3.2. Literature review

The existence of averting or mitigating behaviors was acknowledged earlier in the literature and since the first attempts to provide a consistent theoretical framework of averting behaviors (Courant and Porter, 1981; Harford, 1984; Gerking and Stanley, 1986; Smith and Desvouges, 1986; Bartik, 1988), numerous efforts have been made towards a more thorough understanding of the determinants of these decisions.

The underlying idea is that households undertake averting behaviors to produce a certain level of quality of the environmental goods they consume, or as referred to by Bartik (1988) "the quality of their personal environment". Thus, the decision to perform an environmental defensive behavior is expected to depend on the objective pre-existing quality of environmental conditions faced by consumers. Consequently, many studies use measures of objective quality in order to assess willingness to pay for improvements in environmental quality (Courant and Porter, 1981; Bartik, 1988; Harrington et al. 1989; Abdalla et al. 1992; Laughland et al. 1996). However, consumer judgement about environmental quality and harmful environmental risks has extensively been found to depart from rationality (Slovic, 1987, Simon, 1955, Arrow, 1992, Kahneman 1982). Therefore, conventional models based on objective measures of quality have most of the times been found to be of limited relevance in explaining consumer actual averting choices. Conversely, choice of averting behaviors is expected to be made, instead, on the basis of perceived environmental quality (Um et al. 2002).

As for health risks, they have also been long recognized as one of the main reasons for households undertaking defensive actions. However, given that, as with environmental quality, households may not be perfectly capable of assessing the importance of the risks they are exposed to (Slovic, 1987), subjective measures of health risks are usually employed in empirical works. Similarly, the presence of individuals belonging to vulnerable populations (e.g. young children, elderly people or individuals with poor health status) has been acknowledged to generate risk aversion, sometimes triggering the decision to undertake averting behaviors (Zivin et al 2011; Yoo and Yang, 2000; Abrahams et al. 2000).

With respect to bottled water consumption, these and other factors seem to be affecting the decision to undertake it as an averting behavior. Although some studies have used objective quality of water as a means to assess willingness to pay for an increase in water quality (Harrington et al. 1989; Abdalla et al. 1992; Laughland et al. 1996), perceived measures have been proved to be more relevant in explaining actual behavior (Um et al. 2002). However, evidence on its impact is mixed. While several studies find that households are more prone to consume bottled water the poorer the perceived quality of the water from the tap (Um et al. 2002; Yoo, 2003; Janmaat, 2007), some other find no statistically significant influence of perceived water quality (Larson and Gnedenko 1999; Yoo and Yang, 2000; Doria, 2009). With respect to perceived health risks, evidence is also inconclusive. Some empirical studies conclude that health risk perceptions do have an influence on the averting behaviors undertaken in response to hazardous substances on water (Bontemps and Nauges, 2015; Jakus, 2009). However, others do not find a significant effect (Janmaat, 2007).

Other non-health related aspects of tap water quality are usually found to affect the demand for bottled water as an averting behavior within the household. Those involve mainly organoleptic (aesthetic) characteristics such as taste, odor (mainly chlorine), color and turbidity (i.e the extent to which water has particles in suspension). Research suggests that these sensorial characteristics are at least as important as consumer's health risks perception when deciding on whether or not to undertake averting actions related to drinking water (Abrahams et al. 2000; Jakus et al. 2009). A poor perception about organoleptics has been systematically found to increase the likelihood of the household's choice to consume bottled water (Abrahams et al 2000; Yoo, 2003; Doria, 2009; Jakus et al 2009; Johnstone and Serret, 2012).

Additional consideration would deserve whether households have any previous experience with violations in health-related parameters involving water contamination (Abrahams et al 2000) or past unpleasant episodes with respect to organoleptic characteristics (Dupont et al. 2010, Janmaat, 2007, Um et al 2002). Finally, some aspects related to the perceived quality of the service have also been considered. For instance, Doria (2009) includes satisfaction with tap pressure, finding no significant influence on the propensity to consume bottled water.

With respect to the socio-demographic variables, income is usually considered a determinant. Bottled water is expected to be a normal good, so higher-income households are predicted to both show higher probability of purchasing bottled water (Larson and Gnedenko, 1999; Johnstone and Serret, 2012; Bontemps and Nauges, 2015) and higher level of demand. However, some papers find no significant influence of the income variable (Smith and Desvouges, 1986). Similarly, education is usually included as a proxy for household's knowledge and empirical evidence on its expected sign is mixed. While some find that bottled water consumption increases with education level (Jakus et al. 2009) others do find the opposite effect (Janmaat, 2007) and some elicit no significant relationship (Um et al. 2012). The time that household members have been living in town (Janmaat, 2007; Johnstone and Serret, 2012) and the household size are also usually considered with mixed evidence (Johnstone and Serret, 2012; Yoo and Yang, 2000).

3.3. Model specification

3.3.1. Environmental attitudes and behaviors

Departing from the existing literature, our model addresses several features that have not been sufficiently studied in previous work. First, we analyze the role of both environmental attitudes and behaviors on the decision to undertake averting behaviors that entail negative environmental externalities. Research on the relationship between environmental concerns and behaviors states that environmental concern does not always translate into the corresponding pro-environmental actions (Blake, 1999). This gap between people's attitudes towards the environment and their actual behavior, known as the "value-action gap" or "concern-action paradox," has been long acknowledged and extensively studied (Blake, 1999; Kollmuss and Agyeman, 2002). Furthermore, the literature has identified two differentiated categories of pro-environmental behaviors according to the different levels of sacrifice they impose on the individual, that is, efficiency and curtailment behaviors (Hayes, 1976). Efficiency, or one-shot behaviors refer to the adoption of technologies that conserve certain natural resources (e.g. water or energy) within the household. For example, some efficient behaviors related to water would involve the installation of water-saving devices on taps or the purchase of electrical appliances (i.e. dishwashers or washing machines) that optimize energy consumption. On the other hand, curtailment refers to frequently repeated actions or sacrificial habits that imply a modification in the way people use these resources. Some examples of this type of behaviors include trying to reduce the duration of the showers, or waiting until the dishwasher and washing machine are full before operating them. Therefore, the main difference between both types lies in the fact that while using water-saving technologies does not demand any sacrifice on behalf of the individual, apart from the initial economic cost of installing water-saving devices, having to renounce to a long shower or closing the tap when not in use while brushing teeth entails sacrifices in daily life (Stern and Gardner, 1981).

Although promoting efficient behaviors within the household has usually proved to exhibit more potential for natural resource's conservation than fostering curtailment behaviors (Stern and Gardner, 1981), averting or mitigating actions that we are exploring in this paper are inherently a behavior of the curtailment type. There does not exist an efficient technology that allows households to reduce their consumption without reducing utility. Thus, it is possible that even when environmental concern does not affect consumption habits, individuals that are already undertaking some environmental habits extend it to other pro-environmental behaviors (e.g. restricting their bottled water consumption or using tap water filtration devices). In the same manner, individuals already undertaking a higher level of commitment or sacrifice may find it easier to carry out other behaviors implying similar levels of sacrifice. This has important implications in terms of public policy, since interventions that promote respectively concern, efficiency and curtailment behaviors are rather dissimilar. As for concern, policies normally involve information campaigns aimed at raising household awareness. On the other hand, in the case of behaviors of the one-shot or efficiency class, policy interventions could range from subsides that reduce the cost of purchasing efficient technologies to the design of labeling systems that correctly signal appliance level of efficiency or improving the diffusion of innovation through social networks (Stern and Gardner, 1981; Darley, 1977). Contrary, encouraging the adoption of certain curtailment or sacrificial habits demands a substantially more complex approach involving factors such as generating commitment or leading to changes in social norms (Kollmuss and Agyeman, 2002). As far as averting behaviors are concerned, to our knowledge only one attempt has been made in order to ascertain the impact of environmental concern (Johnstone and Serret, 2012)62 and neither the existence of the abovementioned paradox nor the distinction between different types of behaviors have been previously explored.

3.3.2. Other pricing and non-pricing policies for water conservation.

We explore the influence of several policies for residential water management that may potentially be affecting bottled water demand. These policies can be broadly classified in two categories: pricing and non-pricing (Olmstead and Stavins,

[62] However, this attempt is limited to inclusion of a variable indicating the "how important solid waste issues are for them, relative to other eight environmental concerns" (Johnstone and Serret, 2012: page 674). It only studies the influence of this variable on the propensity to consume bottled water, against both the decision and quantities purchased considered in this study.

2009). Pricing interventions are mainly directed to raising prices and designing tariff schemes that foster efficient water consumption, such as the ones in IBRs (Olmstead et al. 2007). On the other hand, non-pricing instruments include mainly imposing restrictions on water use and promoting efficient water consumption by either influencing individual's attitudes and behaviors or fostering the installation of water saving technologies. In our model, we analyze the influence of both types of policies on the quantity of bottled water consumed.

With respect to pricing, it could be expected that if households are charged a higher price for tap water, they decide to demand more bottled water. It could also be the case that perception of the tap water price[63] by consumers, rather that the actual price, may be affecting bottled water demand. As for non-pricing instruments, the role of individual's attitudes and behaviors have already been discussed previously in this section. Other policies within this category involve water rationing and supply cuts with the objective of reducing demand (Olmstead and Stavins, 2009). Since drinking water is a human necessity, the reliability of the service is very likely to affect the decision to consume bottled water, triggering the need to purchase and store bottled water. Thus, we also aim at exploring the influence of tap water service interruptions.

3.3.3. Model

Our model of bottled water consumption aims to reflect the nature of the decision-making process underlying the decision to purchase bottled water as an averting behavior without making any prior assumption on the process generating it.

As mentioned above, one important issue when dealing with certain averting expenditures, and particularly bottled water consumption, is the high proportion of households that do not consume any amount. Zero consumption may be caused by several reasons. Infrequency of purchase is a usual one. However, given that drinking water is a necessity, it does not seem to apply to the consumption of this particular commodity. The two other common sources, non-participation and corner solutions are more likely to be occurring instead. It may be that some individuals are simply non-consumers of bottled water, that is, that they decide not to "participate" in the market for bottled water for several reasons, but if those reasons were not present (e.g. environmental beliefs explored in this paper), they

[63] Whether or not it is perceived as expensive.

would consume a positive amount. Corner solutions, on the other hand, arise from the consumer's utility-maximizing decision being not to consume at all, given their budget constraint.

This is an important distinction, as econometric modeling strategies will vary according to the economic interpretation placed on those observed zeros. Until now, the majority of existing studies on this issue have focused only on studying the decision or probability to consume bottled water, treating it as a dichotomous variable, without modeling the actual quantity consumed. In the infrequent occasions in which expenditures have been explored, they usually have been investigated through the use of Heckman selection models (Jakus, 2009; Lloyd-Smith et al. 2016) or similar approaches (Yoo and Yang, 2000), thus assuming a priori the non-participation hypothesis. However, the fact that zeros can also arise from corner solutions has been neglected. Within the context discussed above, our model departs from a generic double-hurdle approach (Jones, 1989) in which consumers are presumed to pass two hurdles before observing a positive consumption. First, they decide on whether or not to consume bottled water as an averting behavior and, second, they make a decision on how much bottled water to consume.

Analytically speaking, a representative household is assumed to display a latent utility derived from drinking and using bottled water for household consumption purposes, instead of using water from the tap or other sources (i.e. installing or purchasing water filtration systems). If that utility is positive, they will decide to choose bottled water as an averting behavior, purchasing a certain amount in the market, otherwise they won't.

$$\textit{Participation equation}\colon\ S = f\ (s, e, p, c) \tag{15}$$

Once consumers have decided to consume bottled water, their next decision will be how much water to consume. Thus, the main equation of interest is:

$$\textit{Intensity equation}\colon\qquad Y^* = g(s, e, p, c) \tag{16}$$

where S is a variable reflecting whether or not the household consumes bottled water as opposed to using water from the tap, and Y is the amount of bottled water consumed (quantities purchased). s is a vector of variables including *socioeconomic variables* and variables related to organoleptics and tap water quality, e includes *environmental variables concerning attitudes and behaviors*, p is a vector of *tap water price* related variables, and c accounts for the variables related to *interruptions in the service*. The methodology used to model these decisions and all the variables incorporated in the study are explained in the next section.

Our expectations with respect to the variables, based on previous work, are provided below. A bad perception on both organoleptic and quality is presumed to lead to a higher probability of consuming bottled water, while the effect on the quantities is not clear. Pro-environmental attitudes and behaviors should be expected to reduce the propensity to consume bottled water, and, when the households have decided to consume, based on any other circumstance, it is also expected that environmentally friendly households try to reduce their actual consumption as much as possible. With respect to price, price of tap water and household's perception about that price could be expected to affect the decision on the consumed quantity. Interruptions in the service are expected to affect the decision to consume, since frequent cuts may trigger the need to purchase and store bottled water. The level of disruption caused should be more related to the length in time and other features of those cuts, and could affect both the decision to consume and the quantity to be consumed.

3.4. Data, Variables and Empirical Methodology

3.4.1. Data, Sample and Variables.

We use data from a household survey conducted in the towns of Baza and Guadix, in the province of Granada (southern Spain). Baza and Guadix have a population of 20,668 and 18,928 inhabitants respectively (INE, 2015a), they are located nearly 50 km apart from each other and are served by two different utilities. Water supply originates mainly from snowmelt in the Sierra Nevada mountains, but part of the supply is abstracted from several local springs. In general, objective water quality parameters are rather good and above the official standards.[64] Violations of health related water parameters in this area are rare (only one episode in 2008 is recorded and it was due to torrential rains). However, service interruptions due to network overload are not so uncommon, taking place mainly in the summer, when nearly 28,000 and 23,000 additional tourist residents are added

[64] Values of objective water quality parameters from the last chemical analysis performed can be provided by the authors upon request.

to the regular population of Baza and Guadix respectively (MHAP, 2016), creating excess demand.

The region exhibits certain characteristics that make it an interesting setting for this study. In their last available study about the global market in 2014, the Beverage Marketing Corporation rated Spain as the 4th largest per capita consumer of bottled water in Europe and the 9th in the world in total consumption (Beverage Marketing Company, 2014). Moreover, Spain is a country subject to either water stress or severe water stress throughout the most part of its territory (European Environmental Agency, 2012). Particularly, the towns of Baza and Guadix are located in the Guadalquivir River Basin, a basin under severe water stress (European Environmental Agency, 2012), that has long suffered from water scarcity problems and whose situation is expected to worsen in the future. These circumstances have made water management a paramount issue of concern in the region.

The survey was implemented by a consulting company (Ipsos) in 2014 and administered on a population of 10,062 households in Baza and 9,704 in Guadix (MHAP, 2016), from which a representative sample of 594 households (305 in Baza and 289 in Guadix) was extracted. Sampling was performed with proportional quotas to stratum size, according to gender and age. Questionnaire development included the use of several focus groups and a pilot pre-test. Interviewers were trained before the interview was launched and careful instructions were incorporated into the questionnaire on what information should be conveyed and how responses should be gathered. The survey was administered door-to-door with a response rate of 80%. According to interviewers, respondents had, on average, a very good attitude towards the interview.[65] With respect to the information included in the survey, this database contains a broad set of perceived water quality indicators, as well as usual socioeconomic controls. In order to measure environmental attitudes, the individual had to respond to a series of statements which are able to accurately measure attitudinal factors, from which an aggregate index water built (See Appendix). In addition, a wide range of questions (Appendix 1 in Suárez-Varela and Dinar, 2017) on environmental behaviors is included in order to account for the different levels of individual's environmental involvement (efficiency and curtailment actions). Tap water consumption and prices paid by the household can also be obtained from the questionnaire, and a question on residential water price perception is included. In addition, information on household perception on interruptions of the service was gathered.

[65] They were rated by the interviewers an average of 4.51 in a scale from 1 (Very bad attitude) to 5 (very good attitude)

In order to ascertain our main variables of interest, individuals were asked, respectively, whether or not the household reports using regularly bottled water as main source of drinking and in-house water (e.g. cooking) and the quantity in liters of bottled water consumed per week. It should be noted that when households were asked about bottled water consumption, particular emphasis was made on the fact that we were measuring the use of bottled water as an averting behavior and regular source of drinking water inside the household. Away-from-home or sporadic consumption was not considered. For a more thorough description of the variables, readers are refered to Suárez-Varela and Dinar (2017).

3.4.1.1. Descriptive Statistics

Table 24 depicts the definition and main descriptive statistics of the variables included in the analysis.

Table 24: Descriptive statistics and definition of the variables

Set of variables	*Variable*	*Description*	N	*Mean*	*SD*	*Min*	*Max*
Dependent variables	*Bottledwater*	Household reports consuming bottled water on a regular basis (Dummy)	528	*0.322*	*0.468*	*0*	*1*
	Quantity	Bottled water consumption (in liters per week)	528	*4.333*	*7.748*	*0*	*48*
Socioecon.	*Municipio*	Household is located in Baza (Dummy)	528	*0.496*	*0.501*	*0*	*1*
	HholdIncome	Household income (Ordinal)	528	*6.417*	*3.677*	*1*	*14*
	NoEduc	Respondent has not completed any formal education level (Dummy).	528	*0.047*	*0.213*	*0*	*1*

	BasicEduc	Respondent has completed elementary education(Dummy).	528	*0.348*	*0.477*	*0*	*1*
	Secondary_Educ	Respondent has completed secondary education (Dummy).	528	*0.303*	*0.460*	*0*	*1*
	HighEduc	Respondent has completed university studies either degree, master or PhD (Dummy).	528	*0.301*	*0.459*	*0*	*1*
	Length	Length of time that the respondent has been living in the town (Years).	528	*35.55*	*19.85*	*1*	*86*
	Hsize	Household size (Number of members in the household).	528	*2.955*	*1.157*	*1*	*6*
	Childrenlessthan2	The household reports having members under 2 years old (Dummy).	528	*0.072 0*	*0.259*	*0*	*1*
Water quality and service perception	*Quality*	Satisfaction with water quality 1 (very unsatisfied) - 5 (very satisfied).	528	*4.027*	*1.109*	*1*	*5*
	Serviceperc	Satisfaction with wastewater service. 1 (very unsatisfied) - 5 (very satisfied).	508	*3.415*	*1.178*	*1*	*5*
Organoleptics	*Color*	Respondent perceives that water is not clear 1 (totally disagree) to 5 (totally agree).	528	*1.509*	*0.893*	*1*	*5*
	Smell	Respondent perceives that water has some odor 1 (totally disagree) - 5 (totally agree).	528	*1.555*	*0.878*	*1*	*5*

	Taste	Respondent perceives that water has some taste 1 (totally disagree) - 5 (totally agree).	523	*1.740*	*1.064*	*1*	*5*
Interruptions	*Cutfreq*	Incidence of water supply cuts realized during the summer by the respondent 0 (never) - 5 (very frequently, more than 10 times)	528	*1.246*	*0.508*	*1*	*4*
	Cutdisruption	Supply cuts caused inconvenience to respondent 1 (a few)-5 (a lot).	528	*4.214*	*0.959*	*1*	*5*
Environm. variables	*Envconcernavg*	Respondent average value reported in a set of environmental attitudes (Suárez-Varela and Dinar, 2017.).	528	*3.940*	*0.512*	*1.50*	*5*
	Envworried	Respondent environmental concern is over the mean of the sample (Dummy).	528	*0.540*	*0.499*	*0*	*1*
	Watereff	The household has installed some water saving devices on taps, showers or cisterns (Dummy).	528	*0.616*	*0.487*	*0*	*1*
	filling_dishwasher	Respondent reports waiting until the dish- washer and washing machine are full before operating them (Dummy).	499	*0.972*	*0.165*	*0*	*1*
	Closing_taps	Respondent reports closing the tap while brushing their teeth or shaving (Dummy).	528	*0.936*	*0.246*	*0*	*1*

	Reducing_shower	Respondent reports trying to reduce the duration of his/her shower (Dummy).	528	*0.928*	*0.258 7*	*0*	*1*
	Waterhabitindex	Index indicating number of water conservation habits held by the respondent (Count).	528	*2.78*	*0.48*	*0*	*3*
Price variables	*Priceperception*	Respondent's perception of water tap price 1 (very cheap) – 5 (very expensive).	513	*3.780*	*0.834*	*1*	*5*
	Averageprice	Average price per cubic meter of tap water at the mean point of the range in which the household reports to consume (€/m^3).	394	*1.001*	*0.113*	*0.77*	*1.12*

With respect to our dependent variables, 32.2% of the households report purchasing bottled water on a regular basis. Mean bottled water consumption is 4.33 liters per week, but this mean includes also households that do not consume bottled water at all. Among those households that purchase a positive amount of bottled water, a mean value of of water consumption is 13.7 liters per week. Figure 4 presents the distribution of the consumption values for those households that report consuming bottled water.

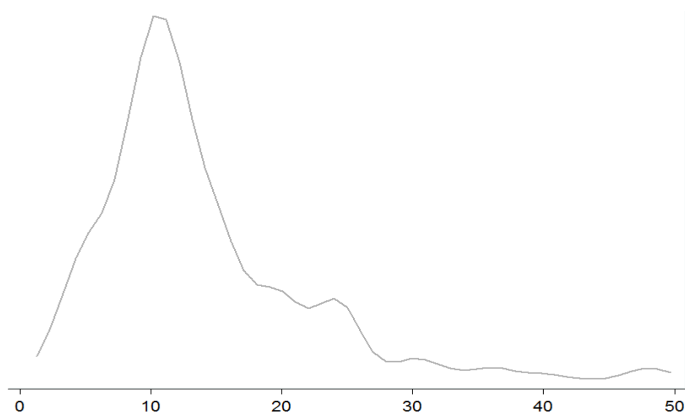


Figure 4: Distribution of bottled water demand in liters per week

Mean household size is 2.95 members, similar to the mean value of 2.51 for Spain (INE, 2015b). 7.2% of the households have at least one child in the household that is younger than 2 years. Mean household income lies within the range of €1,801 - 2,100 per month, slightly lower than the census mean of €2,174 for Spain (INE, 2015b).[66]

With respect to organoleptics, household perception is quite good. In a range of 1 to 5 (5 reflecting poor perception of organoleptics), color has the best perception (1.509); and the worst perception corresponds to taste (1.740). Smell would lie in the middle with a rating of 1.550. In the same vein, consumers are on average more than satisfied with the quality of tap water (4.027 out of 5). As for interruptions in the service, cuts are perceived to be relatively infrequent (between 0 and 3 during the summer), although the level of disruption caused by them is on average high (4.214 out of 5).

The variables related to the environmental value-action gap deserve a more detailed discussion. With respect to environmental attitudes, individuals report, on average, being quite concerned about environmental degradation (3.9 out of 5). In particular, a high percentage of households report practicing water saving habits (curtailment behavior): 97.2% report waiting until the washing machine and dishwasher are full, before operating them; 93.6% try to turn off the tap when not in use while shaving or brushing their teeth, and 92.8% report reducing the duration of their showers. In fact, almost all the households in the survey put into practice at least one of these curtailment behaviors (99.6%).[67] However, as depicted in Figure 5, the level of environmental concern does not seem to make a difference in relation to the involvement in water-saving habits. Those households whose environmental concern is above average have similar values compared with the ones with values below average, and the correlation between environmental concern and the number of water-saving practices that the household performs is substantially low (12.8%).

[66] This is not surprising as Andalucía, the region of Spain where Baza and Guadix are located, is one of the poorest Autonomous Communities in Spain.

[67] This holds even if we restrict more the definition of environmentally concern. For individuals that report more than 4.5 as average environmental concern, these percentages suffer only from little variation (slight increase).

Figure 5: Share of households whose concern is below (light grey) and above (dark grey) average that perform certain water saving behaviors.

With respect to behaviors of the "one-time" or efficiency type, these percentages are relatively lower, with 61.6% of the households in the sample having water-saving technologies installed in the house. Correlation of this variable with the level of environmental concern is also very low (6.6%) and the percentage of households seems to be similar independently of whether the level of concern of the household is above the sample mean or not (Figure 5). As the 'value-action' gap literature asserts, the correlation between environmental concern and behavior in our sample seems weak. In addition, support for the different types of behaviors is substantially diverse, suggesting that the processes generating them may differ. In this context, a further level of disaggregation of the variables related to environmental attitudes and behaviors could improve our understanding of environmental related processes and thus yield more accurate policy recommendations.

3.4.2. Empirical methodology

In this section, we aim to empirically model the demand for bottled water. The first issue that we must deal with is the fact that the sample contains a high percentage of households reporting of not consuming bottled water (67.8%). As referred earlier in the ext, in order to model it, we depart from a generic double-hurdle approach (Jones, 1989) in which consumers are presumed to pass two hurdles before observing a positive consumption. First, they decide on whether or not to consume bottled water (choosing bottled water as their averting behavior) and, once they have decided to consume, they determine the quantity to be consumed. In econometric terms, these decisions can be expressed by the two following equations:

$$\textit{Participation equation}: \; S = \gamma Z \; + \; \nu \tag{17}$$

$$\textit{Intensity equation}: \quad Y^{*} = \beta X + u \tag{18}$$

where ν and u are assumed to have a bivariate normal distribution with zero means, standard deviations σ_u and σ_v, and correlation ρ. Z and X are the covariates affecting each decision explained in the previous section. Because we do not observe utility, instead of S we can only observe whether they have actually participated or not in the market, reflected in a binary choice variable:

$$D = \begin{cases} 1, & s > 0 \\ 0\ , & s \leq 0 \end{cases} \tag{19}$$

As we will explain below, estimation methods will vary according to the assumptions placed on the relationship between the two decisions (joint distribution of the errors) and the process that generates the data (observability rule).

When corner solutions are encountered, values within a certain range are observed as a single value (Greene, 2012). Particularly for the case considered here, when a consumer's underlying utility derived from consuming bottled water is negative ($Y^{*} \leq 0$), the utility-maximizing decision will be not to consume:

$$\textit{Observed consumption}: Y = Y^{\ *} \; \textit{when} \; Y^{*} > 0 \; (D = 1), Y = 0 \; \textit{otherwise} \tag{20}$$

Estimation under this type of censoring of the dependent variable was addressed by Tobin (1958) using a mixture of discrete and continuous distributions. However, one drawback of the Tobit models (as they are usually referred to) is that they estimate only one set of coefficients, implying that the variables in the model affect both the decision to consume and the consumption choice in the same direction. In our setting, this premise may be too restrictive, as there are reasons to believe that the group of factors that influence the choice of bottled water over other averting behaviors related to water consumption are different from the ones that determine the quantity eventually consumed. In order to account for this possibility, we use a more flexible model proposed by Cragg (1971), which allows the participation and intensity equations to be independent and governed by different mechanisms, yielding two different sets of estimations. Thus, in Cragg's models, independence of the disturbance terms (u and v) is assumed ($\rho = 0$) and the participation and consumption equations are estimated respectively by means of a Probit and a truncated regression.

When $\gamma = \beta/\sigma_v$ and provided that the same set of regressors is used for both equations, Cragg's specification will collapse to Tobit (Greene, 2012). A likelihood ratio (LR) test on this restriction proposed by Lin and Schmidt (1984), can be used to choose between Cragg's and Tobit specifications.

On the other hand, when non-participation is suspected as the underlying process generating zero consumption, Heckman selection models are to be applied. In this case, consumption will only be observed when the individuals pass the participation rule ($D = 1$), that is, once they have chosen bottled water as their averting behavior:

$$\textit{Observed consumption}:\ Y = D \cdot Y^{\,*} \tag{21}$$

Under this scenario, the final observed consumption could be biased if there were unobserved factors affecting both the decision to consume and the quantity actually consumed. Therefore, under Heckman models, dependence of the disturbance terms (u and v) is presumed in order to account and correct for the possibility of the existence of selection bias. Parameters in the system can be estimated through either Full Information Maximum Likelihood (FILM) or two-step estimation (Heckman, 1979) and, after estimation, the independence assumption can be tested by means of a LR test. In the event that both errors were found to be correlated, the existence of selection bias would be uncovered in our sample, and OLS would be yielding inconsistent estimates. However, in the case of $\rho = 0$, independence of the two decisions can be assumed and two-part models in which a Probit and OLS equations are estimated separately for each decision, have proved more efficient. Moreover, when $\rho = 0$, a Vuong test for non-nested models to test for the truncated normal against the lognormal specifications can be applied to discern between the Cragg's and Heckman[68] specifications (Wooldridge, 2010).

In addition, when using Heckman selection models, in order for the system to be properly identified, Z must contain at least one regressor, also known as exclusion restriction, that must belong to the participation equation while being exogenous to the consumption decision, and thus not included in X.

Finally, in order to determine the magnitude of the response of the variable of interest under a change in one of the independent variables, marginal effects should be obtained. Here, we are interested in predicting unconditional marginal effects, that is, the potential change in bottled water consumption that could be achieved through a public policy affecting one of the independent variables. In the case of Heckman models, unconditional partial effects can be interpreted directly from the estimation results. However, in Cragg's approach, obtaining unconditional marginal effects requires some extra calculations of marginal impacts:

$$\begin{aligned}\frac{\partial E[y|Z,X]}{\partial x_j} &= \gamma_j \phi(Z\gamma)\left[X\beta + \sigma\lambda\left(\frac{X\beta}{\sigma}\right)\right] \\ &\quad + \Phi(Z\gamma)\beta_j\left[1-\lambda\left(\frac{X\beta}{\sigma}\right)\left\{\frac{X\beta}{\sigma}+\lambda\left(\frac{X\beta}{\sigma}\right)\right\}\right]\end{aligned} \tag{22}$$

where ϕ is the normal density function, Φ is the normal distribution function and $\lambda\left(\frac{X\beta}{\sigma}\right) = \phi\left(\frac{X\beta}{\sigma}\right) / \Phi\left(\frac{X\beta}{\sigma}\right)$ is the Inverse Mills Ratio (IMR).

3.4.2.1. Endogeneity

Another issue that should be addressed is that in our model two variables are suspected of being endogenous: the index on water saving habits and the price structure. Water saving habits could be expected to be jointly determined with bottled water consumption if there are individual unobservable characteristics that foster both the decision to consume bottled water and to reduce tap water consumption by introducing certain saving habits. With respect to the price variable, the municipalities in our sample apply tariffs for tap water that takes the form of Increasing Block Rates (IBRs). In block pricing, simultaneity bias is expected to arise from the fact that price increases with the quantity consumed,

[68] This is true when log(y) is effectively treated as the dependent variable.

which in turn is affected by prices (Olmstead and Stavins, 2007). Because bottled water demand is closely related to the household decision on tap water consumption, it is to be expected that tap water price will be endogenous to the decision on bottled water consumption.

In order to account for endogeneity in the framework of selection models, Wooldridge (2010) proposes a two-step approach in which a probit model is estimated for the selection indicator, including all exogenous variables (i.e. instruments for the endogenous regressor, exogenous regressors in the intensity equation and exclusion restrictions) and then the IMR is computed and included in a 2SLS estimate of the structural equation (equation of interest). Because standard errors are incorrect when the IMR coefficient is statistically different from zero, bootstrapping should be applied (Wooldridge, 2010).

For corner solution models (Tobit and Cragg's), a control function approach is used. In a first step, the endogenous variable is regressed on the exogenous regressors and the set of instruments and, after estimation, the residuals are retrieved. Estimated residuals are included in the models' equations. The inclusion of this error term in the equations of interest corrects for endogeneity, the test for the significance of the error term becomes a test for endogeneity. Similarly as in the case of selection models, the inclusion of a generated regressor created in a previous estimation is addressed using bootstrapping.

A final issue is finding valid and relevant instruments. As common practice, we use the set of all the possible marginal prices for each block as instruments for average price of tap water (Olmstead, 2009). With respect to the index reflecting water habits, we use several questions reflecting household's concern and willingness to act related particularly with efficient and sustainable use of water resources and supply networks (see Appendix). These variables are expected to be correlated with the household's decision on performing water saving habits while not affecting demand for bottled water.

3.5. Results

Results of the different estimated models are reported in Table 25. Since some households did not report their water bill, the variable *averageprice* suffers from a significant number of missing values (around 20%). For that reason, we first estimate a model with all the variables excluding *averageprice*, and then we model the relationship with the price variable (Table 30).

Table 25: Heckman selection (FILM), Two-part, Tobit and Cragg's model estimations (N=493; Censored= 332).

	Heckman		OLS	Tobit	Cragg	
VARIABLES	Participation	Intensity			Participation	Intensity
Municipio	-0.225	0.0339	0.0816	-1.776	-0.228	2.518
	(0.146)	(0.100)	(0.0965)	(1.996)	(0.145)	(1.709)
Childrenlessthan2	0.455*		-0.0669	4.160	0.435*	-0.643
	(0.245)		(0.135)	(3.216)	(0.254)	(2.362)
Length	-0.0108***	-0.00425	-0.00257	-0.150***	-0.0109***	-0.0288
	(0.00380)	(0.00269)	(0.00250)	(0.0527)	(0.00379)	(0.0450)
Hholdincome	-0.00687	-0.00907	-0.00877	-0.230	-0.00707	-0.357
	(0.0214)	(0.0130)	(0.0135)	(0.284)	(0.0209)	(0.246)
Hsize	-0.0417	0.185***	0.187***	0.817	-0.0360	3.405***
	(0.0600)	(0.0365)	(0.0380)	(0.815)	(0.0596)	(0.720)
BasicEduc	0.0254	0.129	0.167	-0.0632	-0.0105	3.283
	(0.363)	(0.263)	(0.271)	(5.115)	(0.359)	(5.076)
SeconEduc	0.170	0.251	0.253	3.003	0.154	5.256
	(0.374)	(0.276)	(0.289)	(5.305)	(0.370)	(5.341)
Higheducation	0.0818	0.104	0.130	1.624	0.0635	3.263
	(0.386)	(0.277)	(0.286)	(5.415)	(0.380)	(5.315)
Color	0.159	-0.00158	-0.0255	1.862	0.148	-0.796
	(0.0970)	(0.0610)	(0.0600)	(1.279)	(0.0963)	(1.044)
Smell	0.0842	0.00276	-0.00382	1.047	0.0872	0.729
	(0.0989)	(0.0536)	(0.0548)	(1.264)	(0.0955)	(0.965)
Taste	0.141*	0.0768*	0.0518	2.144**	0.121	1.148
	(0.0780)	(0.0433)	(0.0406)	(0.990)	(0.0753)	(0.718)

Quality	-0.317***	-0.112**	-0.0558	-3.985***	-0.315***	-0.931
	(0.0769)	(0.0550)	(0.0400)	(0.983)	(0.0761)	(0.718)
Serviceperc	0.102	0.00571	-0.0186	1.117	0.114*	-0.328
	(0.0647)	(0.0410)	(0.0373)	(0.850)	(0.0649)	(0.666)
Cutfreq	0.346**	0.0241	-0.0271	3.813**	0.381***	-0.760
	(0.148)	(0.0867)	(0.0784)	(1.844)	(0.147)	(1.376)
Cutdisruption	0.0525	-0.00908	-0.0217	0.488	0.0541	-0.223
	(0.0733)	(0.0439)	(0.0441)	(0.987)	(0.0731)	(0.813)
Envconcernavg	0.305	0.118	0.0433	3.925	0.303	0.250
	(0.255)	(0.180)	(0.177)	(3.458)	(0.246)	(3.169)
Envworried	-0.120	0.0361	0.0761	-0.373	-0.115	2.083
	(0.231)	(0.152)	(0.157)	(3.156)	(0.227)	(2.803)
Watereff	-0.0427	-0.000460	-0.000347	-0.123	-0.0465	0.226
	(0.148)	(0.0983)	(0.102)	(2.043)	(0.148)	(1.827)
Waterhabitindex	-0.429**	-0.361**	-0.299**	-7.319**	-0.422**	-6.192**
	(0.208)	(0.144)	(0.143)	(2.852)	(0.206)	(2.599)
Priceperception	-0.0100	0.110**	0.118**	0.636	-0.00102	1.746*
	(0.0860)	(0.0522)	(0.0536)	(1.161)	(0.0853)	(0.969)
Constant	-1.363	1.006	1.527*	-23.75	-1.433	-4.188
	(0.276)	(0.272)	1.527*	15.51***	(1.213)	(15.60)
ρ	0.614					
	(0.439)					
Σ	-0.672***				7.779963***	
	(0.140)				(0.5984736)	
LR test of independent equations	$\chi^2_1 = 1.08$ (0.2977) [a]					

Standard errors in parentheses.

Notes: *, ** and *** denote 10%, 5%, and 1% significance levels, respectively.

[a] P-value.

First, models with endogenity correction for *waterhabitindex* were run. Tests for validity, relevance of the instruments and endogeneity are reported in Table 26. In the Heckman model, since the second stage is a 2SLS, validity and relevance of the instruments are confirmed by a Sargan test of overidentifying restrictions and an F-test of excluded instruments (Bound, Jaeger and Baker, 1995) respectively. However, Hausman test for endogeneity fails to be rejected, indicating that there is no need to instrument. In the case of the Cragg's model, as proposed by Wooldridge (2010), an F-test of exclusion of instruments is performed,[69] confirming instruments' validity. Moreover, an F-test on the first stage regression indicates also relevance. Nevertheless, the t-test on the coefficient of the estimated residual is not rejected, also pointing out to endogeneity correction for this variable not being necessary in the Cragg's specification. Thus, models without endogeneity correction are finally performed.

[69] After running the structural equation with the control function (residual from the first stage) included, instrumental variables should not belong to the structural equation. Under that logic, the structural equation with endogeneity correction is run (including all instruments except for one) and an F-test on those instruments is conducted. In order for those instruments to be valid, they should not be jointly significant in an F-test of exclusion of instruments. The test is invariant to the choice of excluded instrument (Wooldridge, 2010)

Table 26: Tests for endogeneity, validity and relevance for Heckman selection (FILM) and Cragg's model estimations with endogeneity correction for the variable waterhabitindex.

Tests	Heckman	Tests	Cragg	
			Participation	Intensity
Hausman	0.02 (0.8951)	T-test on the included residual	-0.44 (0.663)	1.13 (0.257)
Sargan test	0.1914 (0.6618)	F-test of exclusion of instruments	0.761 (0.6835)	0.743 (0.6897)
F-test of excluded instruments (First stage 2SLS)	15.50 (0.0014)	F-test (First stage)	3.00 (0.0305)	

p-values are reported in parentheses.

In the Heckman specification, *Childrenlessthan2* is used as an exclusion restriction. Having children less than two years old has been found to impact the likelihood of purchasing bottled water (see literature review), but it should not necessarily affect the amount finally consumed. As expected, Table 25 shows that it is a significant determinant on the decision to consume (participation equation in Heckman model), while not affecting the quantity consumed in a separate OLS estimate of the intensity equation, thus posing an adequate exclusion restriction.

Heckman model yields a ρ of 0.614. However, a direct test for the existence of the selection effect ($\rho = 0$) cannot be rejected, implying independent errors. A likelihood ratio test for the independency of both equations also cannot be rejected, favoring the estimation of a separate probit model for the participation equation and a regression model on the intensity decision against the Heckman specification.

Results for the Tobit and Cragg's model are also reported (Table 25). A likehood ratio test (Lin and Schmidt, 1984) for the restriction of Tobit model yields a value of 28.7, rejecting the null hypothesis that $\gamma = \beta/\sigma_v$ at a 1% level, and thus favoring Cragg's more flexible specification against Tobit. Finally, a Vuong test for non-nested models to compare the lognormal and truncated specifications is performed. With a value of -0.146 and a p-value of 0.010, the Vuong test is rejected at the 1% level, implying that Cragg's model should be preferred to its Heckman's counterpart, and thus favoring the hypothesis of corner solution being the process governing observed zero consumption. Therefore, Cragg's specification will be our final modeling choice. In any case, results are found to be very robust across the various econometric specifications (See Table 25).

In order to study the magnitude of the effect of those variables on both the probability to consume and the quantity of bottled water consumed, marginal effects are computed. For the intensity equation, we report unconditional marginal effects (Table 27) accounting for the total potential effect (that is, both the direct effect on quantity and the indirect effect through the change in the probability to consume) on bottled water consumption that could be achieved through a change in each of the independent variables. For the standard errors to be valid, we estimate them using bootstrapping (Wooldridge, 2010).

Table 27: Marginal Effects for the Cragg's model estimations.

VARIABLES	Marginal effects	
	Participation	Intensity (Unconditional)
Municipio	-0.0638819	-0.2073927
	(0.0404749)	(0.7594287)
Childrenlessthan2	0.1219732*	1.417454
	(0.0651619)	(1.030329)
Length	-0.0030683***	-0.0466816***
	(0.0009935)	(0.0180355)
Hholdincome	-0.0019846	-0.1132304
	(0.0054422)	(0.101826)
Hsize	-0.0100926	0.7046737**
	(0.0171237)	(0.3108281)
BasicEduc	-0.0029481	0.7668188
	(0.0863704)	(2.175619)
SeconEduc	0.043138	1.845703
	(0.1146203)	(2.508469)
Higheducation	0.0178282	1.030414
	(0.1038933)	(2.320931)
Color	0.0416047	0.3420471
	(0.0288384)	(0.5051409)
Smell	0.0244638	0.4946904
	(0.0303447)	(0.56174)
Taste	0.0340424	0.7211814**
	(0.0268369)	(0.3493412)
Quality	-0.0883292***	-1.368968
	(0.0297569)	(0.3534333)

Serviceperc	0.0318698	0.3311705
	(0.0207237)	(0.2342998)
Cutfreq	0.1069017*	1.194052
	(0.0591058)	(0.8170846)
Cutdisruption	0.0151802	0.1412496
	(0.0268184	(0.3591169)
Envconcernavg	0.0849113	1.157794
	(0.0731812)	(1.155939)
Envworried	-0.0321705	0.0954011
	(0.0640323)	(0.9549162)
Watereff	-0.0130449	-0.1129135
	(0.0382911)	(0.617297)
Waterhabitindex	-0.1182606**	-3.045422***
	(0.0487385)	(1.009367)
Priceperception	-0.0002867	0.4243339
	(0.0262441)	(0.4545721)

Standard errors in parenthesis are computed using bootstrapping with 100 iterations. Notes: *, ** and *** denote significance at 10%, 5%, and 1% significance levels, respectively.

We find that, as expected, some factors such as lower quality perception of the water from the tap increase the probability of drinking bottled water, while not affecting the quantity eventually consumed. This suggests that characteristics related to water quality tend to affect the decision to use bottled water more than the amount consumed, once the individual decides to purchase bottled water as an averting behavior. Although perceived taste is not significant in any of the equations when estimated separately, the joint marginal effect on quantity is significant, suggesting that targeting this variable by policy intervention is also expected to affect bottled water consumption.

We also find that households with children younger than two years report a higher probability (12.2%) of choosing to consume bottled water. This result is in line with previous literature (Yoo and Yang, 2000) and seems to indicate that when households display a higher level of risk aversion, their propensity to consume bottled water is also higher. The length that the consumer has been living in the same town seems to lead to a decreased probability of purchasing bottled water. This is usually explained by the fact that familiarity leads to a reduction of risk perception (Slovic, 2000) and with time people get accustomed to the organoleptic characteristics of tap water (Doria, 2010). Quantity, however, seems to be better explained by household size, that is, as expected, bottled water consumption is predicted to increase with the number of members in the household. Finally, in relation to the perception of the quality of the service, contrary to what was expected a higher satisfaction with wastewater treatment is associated with a higher probability of consuming bottled water.

With respect to the variables related to interruptions in the service, a perception by the household that supply cuts are more frequent is shown to lead to a higher probability to purchase bottled water, while not affecting the level of consumption. A marginal increase in this indicator, while holding all other indicators constant, is expected to increase the probability of purchasing bottled water by up to 10.7%. This result suggests that, as we expected, service interruptions could generate a need to keep bottled water to prevent lack of water supply. However, the length and frequency of disruption doesn't seem to affect neither the probability to consume nor the quantity of bottled water to be consumed.

As for the analysis of the environmental paradox, our results show that environmental concern does not translate into a reduction in either the probability to consume bottled water nor the quantity consumed. Likewise, the fact that individuals carry on behaviors of the "efficient" type, that is, one-time behaviors such as installing certain types of water saving devices, does not seem to affect bottled water consumption or the level of consumption. However, we do observe that those individuals that consistently undertake a higher number of daily saving habits, also show both a lower probability of choosing to consume bottled water and a lower quantity consumed. Thus, our results seem to suggest that individuals showing a higher level of commitment towards environmental degradation in their daily lives are more prone to carry out other behaviors entailing similar levels of sacrifice in order to reduce their environmental impact. Moreover, the joint effect of this variable is found to be the most sizeable one, with a marginal increase in this indicator, while holding all other indicators constant, predicted to reduce consumption by up to a 22% of the current average consumption per week exhibited by the households in the sample.

Finally, price perception is found to affect bottled water consumption as well. Our results suggest that households that do perceive tap water as more expensive tend to consume more bottled water. Since it has been proved (Nieswiadomy and Molina, 1989; De Bartolome, 1995) that consumers have problems in understanding water tariffs, it may be that if water from the tap is perceived more expensive, consumers assess the relative cost of diverting to drinking bottled water as being smaller.

3.5.1. Robustness Checks

In order to show the robustness of the estimations, Table 28 and Table 29 display respectively *step-wise estimations* by groups of variables for the participation and intensity equations of our final model choice (Cragg's Tobit specification). Moreover, in a previous section, robustness *across different methodological specifications* was also shown.

Table 28: Robustness checks. Participation equation of the final chosen model (Cragg's Tobit)

VARIABLES	Model 1 (Socioecon.)	Model 2 (+ Water quality)	Model 3 (+ Interruptions)	Model 4 (+Environ.)	Model 5 (+Price perception)
Municipio	-0.333***	-0.322**	-0.293**	-0.247*	-0.228
	(0.120)	(0.134)	(0.136)	(0.143)	(0.145)
Childrenlessthan2	0.438**	0.538**	0.520**	0.462*	0.435*
	(0.223)	(0.246)	(0.250)	(0.249)	(0.254)
Length	-0.0125***	-0.0102***	-0.00995***	-0.0106***	-0.0109***
	(0.00328)	(0.00364)	(0.00366)	(0.00373)	(0.00379)
Hholdincome	-0.0294*	-0.0222	-0.0114	-0.00631	-0.00707
	(0.0172)	(0.0191)	(0.0198)	(0.0206)	(0.0209)
Hsize	-0.0124	-0.0332	-0.0471	-0.0422	-0.0360
	(0.0520)	(0.0573)	(0.0580)	(0.0592)	(0.0596)
BasicEduc	0.0653	0.0779	0.0847	-0.0209	-0.0105
	(0.315)	(0.357)	(0.358)	(0.356)	(0.359)
SeconEduc	0.217	0.231	0.225	0.112	0.154
	(0.325)	(0.365)	(0.368)	(0.366)	(0.370)
Higheducation	0.327	0.215	0.234	0.0861	0.0635
	(0.331)	(0.373)	(0.375)	(0.374)	(0.380)
Color		0.167*	0.144	0.153	0.148
		(0.0916)	(0.0936)	(0.0954)	(0.0963)
Smell		0.0717	0.0633	0.0638	0.0872
		(0.0933)	(0.0937)	(0.0938)	(0.0955)
Taste		0.134*	0.126*	0.124*	0.121
		(0.0737)	(0.0738)	(0.0749)	(0.0753)

Quality		-0.322***	-0.319***	-0.311***	-0.315***
		(0.0737)	(0.0744)	(0.0758)	(0.0761)
Serviceperc		0.0981	0.118*	0.116*	0.114*
		(0.0615)	(0.0628)	(0.0637)	(0.0649)
Cutfreq			0.289**	0.331**	0.381***
			(0.140)	(0.145)	(0.147)
Cutdisruption			0.0739	0.0674	0.0541
			(0.0705)	(0.0718)	(0.0731)
Envconcernavg				0.348	0.303
				(0.243)	(0.246)
Envworried				-0.130	-0.115
				(0.223)	(0.227)
Watereff				-0.0909	-0.0465
				(0.147)	(0.148)
Waterhabitindex				-0.344*	-0.422**
				(0.193)	(0.206)
Priceperception					-0.00102
					(0.0853)
Constant	0.108	0.379	-0.368	-1.576	-1.433
	(0.368)	(0.559)	(0.706)	(1.159)	(1.213)
Observations	528	503	503	503	493
Log-likelihood	-865.12747	-783.29806	-780.59767	-772.90769	-762.65091
Sigma	8.2275***	8.1389***	8.1170***	7.8541***	7.7799***
	(0.63770)	(0.6430)	(0.63992)	(0.60618)	(0.59847)

Standard errors in parentheses.

Notes: *, ** and *** denote 10%, 5%, and 1% significance levels, respectively.

Reported likelihood refers to the joint estimation of the two equations in the model (Probit and truncated regression).

Table 29: Robustness checks. Intensity equation of the final chosen model (Cragg's Tobit).

VARIABLES	Model 1 (Socioeconomic)	Model 2 (+ Water quality)	Model 3 (+ Interruptions)	Model 4 (+Environment)	Model 5 (+Price perception)
Municipio	3.589**	3.754**	3.383**	2.769	2.518
	(1.580)	(1.621)	(1.674)	(1.716)	(1.709)
Childrenlessthan2	-1.399	-1.125	-0.805	-1.068	-0.643
	(2.364)	(2.421)	(2.439)	(2.367)	(2.362)
Length	-0.0395	-0.0194	-0.0158	-0.0236	-0.0288
	(0.0429)	(0.0448)	(0.0448)	(0.0447)	(0.0450)
Hholdincome	-0.158	-0.267	-0.290	-0.294	-0.357
	(0.222)	(0.243)	(0.245)	(0.244)	(0.246)
Hsize	3.296***	3.298***	3.388***	3.623***	3.405***
	(0.695)	(0.728)	(0.738)	(0.723)	(0.720)
BasicEduc	2.854	4.804	4.353	2.756	3.283
	(5.026)	(5.283)	(5.300)	(5.107)	(5.076)
SeconEduc	3.051	5.900	5.680	3.920	5.256
	(5.115)	(5.457)	(5.488)	(5.328)	(5.341)
Higheducation	2.113	3.831	3.382	1.336	3.263
	(5.137)	(5.386)	(5.392)	(5.230)	(5.315)
Color		-0.542	-0.396	-0.569	-0.796
		(1.054)	(1.063)	(1.051)	(1.044)
Smell		0.572	0.469	0.756	0.729
		(0.987)	(1.001)	(0.982)	(0.965)
Taste		0.760	0.777	1.129	1.148
		(0.723)	(0.721)	(0.723)	(0.718)

Quality		-0.999	-1.033	-1.219*	-0.931
		(0.733)	(0.734)	(0.711)	(0.718)
Serviceperc		-0.333	-0.250	-0.321	-0.328
		(0.691)	(0.701)	(0.674)	(0.666)
Cutfreq			-1.146	-1.122	-0.760
			(1.381)	(1.380)	(1.376)
Cutdisruption			0.138	-0.102	-0.223
			(0.809)	(0.819)	(0.813)
Envconcernavg				-0.414	0.250
				(3.192)	(3.169)
Envworried				2.842	2.083
				(2.804)	(2.803)
Watereff				-0.0183	0.226
				(1.827)	(1.827)
Waterhabitindex				-6.108**	-6.192**
				(2.530)	(2.599)
Priceperception					1.746*
					(0.969)
Constant	0.520	1.048	5.267	5.267	-4.188
	(5.675)	(7.076)	(14.83)	(14.83)	(15.60)
Observations	528	503	503	503	493

Standard errors in parentheses.

Notes: *, ** and *** denote 10%, 5%, and 1% significance levels, respectively.

3.5.2. Models with Tap Water Price

Results of the models including the price variable (*averageprice*) and addressing its likely endogeneity are reported in Table 30. The estimated coefficient for the Inverse Mills Ratio is negative and significant in the intensity equation of the selection model, suggesting that the selection effect should be accounted for. In the presence of selection bias, independence of the disturbance terms should not be assumed, and therefore Heckman's specification should be preferred to its Cragg's counterpart.

As for endogeneity of the price variable, for it to be corrected the set of instruments must be valid and relevant. Because in a second step of the selection model we use 2SLS, a Sargan test of overidentifying restrictions is performed. With a value of 0.26316 (p=0.8767), Sargan test cannot be rejected, indicating that the set of instruments is valid. Results for the first stage estimations are also included in Table 30. Following Bound, Jaeger and Baker (1995), relevance of the instruments is confirmed by the rejection of the F-test of excluded instruments on the set of instrumental variables. Finally, a Hausman test for endogeneity is also rejected at the 10% level, recommending the use of endogeneity correction.

With respect to our variable of interest, our results show that bottled water demand reacts to the price of tap water. In order to understand the magnitude of the effect, elasticities are computed from the estimated coefficients, yielding a positive and significant cross-price elasticity of *10.72*. That is, a 1% increase in the average price for tap water is expected to increase bottled water demand by up to 10.72%, implying that bottled and tap water would be substitute goods.

Table 30: Estimates of Heckman selection model with IV and Cragg's model with control function approach to correct for price endogeneity (N=379. Censored= 252).

	Heckman with IV			Cragg with CFA		
VARIABLES	Participation	First stage 2SLS	Intensity 2SLS	First stage	Participation	Intensity
Municipio	4.778	0.129	-6.445**	0.110	-0.788	-13.86
	(699.5)	(0.164)	(2.752)	(0.0876)	(0.738)	(9.848)
Childrenlessthan2	0.678**				0.654**	0.782
	(0.287)				(0.288)	(2.722)
Length	-0.0167***	1.72e-05	-0.0628**	7.77e-05	-0.0162***	-0.0417
	(0.00430)	(0.000538)	(0.0274)	(0.0003)	(0.00429)	(0.0401)
Hholdincome	-0.0246	-0.000307	-0.126	-0.000227	-0.0117	-0.388*
	(0.0256)	(0.00184)	(0.125)	(0.00156)	(0.0252)	(0.221)
Hsize	-0.0787	0.00933*	-0.144	0.00953**	-0.165*	1.270
	(0.0731)	(0.00541)	(0.372)	(0.00445)	(0.0920)	(1.000)
BasicEduc	0.217	-0.0397**	3.814**	-0.0405	0.339	17.87**
	(0.496)	(0.0194)	(1.845)	(0.0279)	(0.551)	(6.939)
SeconEduc	0.213	-0.0395**	4.441**	-0.0402	0.392	19.18***
	(0.507)	(0.0195)	(1.968)	(0.0288)	(0.556)	(6.938)
Higheducation	0.211	-0.0290	3.267*	-0.0297	0.315	14.43**
	(0.516)	(0.0214)	(1.776)	(0.0295)	(0.550)	(6.638)
Color	0.0935	0.000245	0.514	1.02e-05	0.0960	-1.924*
	(0.117)	(0.00828)	(0.826)	(0.00757)	(0.117)	(1.065)
Smell	0.150	0.00681	-0.0666	0.00625	0.105	0.870
	(0.111)	(0.00753)	(0.963)	(0.00733)	(0.113)	(0.992)
Taste	0.0922	0.00292	0.126	0.00261	0.0905	0.628

	(0.0868)	(0.00402)	(0.631)	(0.00575)	(0.0870)	(0.661)
Quality	-0.315***	-0.0133	-1.068*	-0.0124**	-0.302***	0.000395
	(0.0892)	(0.0105)	(0.595)	(0.00560)	(0.117)	(1.133)
Serviceperc	0.0829	0.00154	0.369	0.00130	0.0973	-0.707
	(0.0760)	(0.00542)	(0.385)	(0.00466)	(0.0763)	(0.595)
Cutfreq	0.357**	0.0122	-0.00759	0.0110	0.286*	-2.057
	(0.170)	(0.0144)	(0.930)	(0.0103)	(0.164)	(1.378)
Cutdisruption	0.0938	0.0102	-0.410	0.00988*	0.0384	-1.851*
	(0.0879)	(0.00625)	(0.500)	(0.00535)	(0.0984)	(1.013)
Envconcernavg	0.320	0.0182	-1.294	0.0169	0.226	-1.896
	(0.310)	(0.0241)	(1.440)	(0.0182)	(0.325)	(3.193)
Envworried	-0.0771	0.0246	-0.0613	0.0248	-0.213	0.698
	(0.276)	(0.0195)	(1.411)	(0.0169)	(0.300)	(3.215)
Watereff	0.00535	0.0111	-0.747	0.0110	-0.0332	-2.612
	(0.177)	(0.0133)	(0.878)	(0.0106)	(0.184)	(1.769)
Waterhabitindex	-0.390	-0.0265	0.316	-0.0250*	-0.333	0.173
	(0.267)	(0.0204)	(1.303)	(0.0135)	(0.305)	(3.708)
Priceperception	-0.0210	-0.000676	0.679	-0.000395	0.0277	2.140**
	(0.0993)	(0.00870)	(0.535)	(0.00607)	(0.0984)	(0.834)
Averageprice	-0.0854		47.01**		3.508	112.5
	(0.943)		(20.83)		(5.747)	(79.56)
IV1	-57.66	-1.155		-0.936		
	(7,972)	(1.864)		(1.081)		
IV2	17.99	-0.0633		-0.141		
	(1,056)	(0.634)		(0.131)		
IV3	3.512*	0.273**		0.261***		
	(1.901)	(0.114)		(0.0974)		
IMR		0.00490	-1.202***			
		(0.0407)	(0.340)			

Residual					-4.399	-97.18
					(5.883)	(78.44)
Constant	-5.010	0.841***	-31.30*	0.8615***	-4.537	-95.47
	(1,015)	(0.248)	(17.82)	(0.143155 3)	(5.206)	(68.12)
Σ					6.280***	
					(0.488)	
Sargan test		$\chi^2_2 = 0.26316$				
		(0.8767)				
F-test of excluded instruments		$F_{(3,355)} = 2.78$				
		(0.041)				
Hausman test.		$\chi^2_1 = 3.40$				
.		(0.0651)				

Standard errors in parenthesis are computed using bootstrapping with 100 iterations.

Notes: *, ** and *** denote 10%, 5%, and 1% significance levels, respectively.

3.6. Conclusions

Households tend to adopt averting behaviors in response to potential environmental hazards or an insufficient quality of their environmental conditions. Many of these averting behaviors, besides implying costs for individuals and societies, also lead to environmental externalities. Particularly, the undertaking of certain mitigating behaviors is expected to increase in the following decades with significant negative consequences for the environment (Estrada et al., 2017; Isaac and Van Vuuren, 2009). In order to design public policies that are able to contain the externalities, we want to explore the role of individual's environmental attitudes and behaviors in the choice of averting behaviors that pose environmental negative externalities. An econometric strategy is proposed in order to deal with the substantial proportion of zero responses usually found in empirical studies on averting behaviors.

Using a dataset on bottled water consumption from two cities in southern Spain facing severe water scarcity, we further extend the analysis to explore the impact of some public policies related to residential water management. Our results reveal that neither environmental concern nor behaviors of the one-shot type are predictors of a reduced bottled water consumption. However, those individuals that more consistently maintain behaviors of curtailment type seem to show both a lower probability to divert to bottled water and lower levels of consumption, with the magnitude of this effect being the most sizable one among the variables considered in our study. We also find that a perception by the household on interruptions being more frequent increase the probability of diverting to bottled water consumption, while higher price paid for the water from the tap as well as the perception that it is more expensive are related to higher levels of consumption, with cross-price elasticity of bottled and tap water being positive and significant. We realize that some of the distributional methodological assumptions previously imposed in the literature prove to be restrictive and not always supported by our data.

Therefore, our results suggest that public policies aimed at promoting environmental habits can prove very successful in containing averting behaviors that display substantial environmental negative externalities. They seem to indicate that some pricing and non-pricing policies related to the efficient management of water resources may result in environmentally undesirable effects derived from an increase in bottled water demand. Consequently, accounting for those environmental costs seems necessary for an accurate assessment of the environmental effects of certain water conservation policies. Likewise, an important conclusion is that, when the number of households that do not undertake the averting behavior is significant, special attention should be paid to the modeling strategy, as improper modeling of zero consumption would lead to misleading conclusions.

Appendix: Set of questions used as instruments for the index on water-saving habits.

1. Do you think that, as it is proposed by EU Norms, your municipality should take steps towards a more efficient and sustainable use of water resources and, particularly, towards reducing network losses? [Yes/No]
2. Do you have an approximate idea about the percentage of water network losses in your municipality? [Yes/No]
3. Would you be willing to pay an extra amount in your water bill to act more decisively in order to improve the current state of the supply networks? [Yes/No]

References

Abdalla, C.W., Roach, B.A. and Epp, D.J. (1992). Valuing environmental quality changes using averting expenditures: An application to groundwater contamination. Land Economics 68(2): 163-169.

Abrahams, N.A., Hubbell, J. and Jordan, J.L. (2000). Joint production and averting expenditure measures of willingness to pay: Do water expenditures really measure avoidance costs?. American Journal of Agricultural Economics 82(2): 427-437.

Arrow, K. J. (1982). Risk Perception in Psychology and Economics. Economic Inquiry 20: 1–9.

Arnold, E. and Larsen, J. (2006). Bottled Water: Pouring Resources Down the Drain. Earth Policy Institute (EPI), Plan B Updates. Washington, DC. Available at: http://www.earth-policy.org/plan_b_updates/2006/update51 (Available January 12, 2017).

Bartik, T. J. (1988). Evaluating the Benefits of Non-Marginal Reductions in Pollution Using Information on Defensive Expenditures. Journal of Environmental Economics and Management 15: 111–127.

Beverage Marketing Corporation (2016). U.S. Bottled water through 2020. Beverage Marketing Corporation, New York.

Beverage Marketing Corporation (2014). The Global Bottled Water Market. Beverage Marketing Corporation, New York.

Blake, J. (1999). Overcoming the 'value-action gap' in environmental policy: tensions between national policy and local experience. Local Environment: The International Journal of Justice and Sustainability, 4(3): 257–278.

Bontemps, C. and Naugues, C., (2015). The Impact of Perceptions in Averting-decision Models: An Application to the Special Regressor Method to Drinking Water Choices. Americal Journal of Agricultural Economics 98 (1): 297-313.

Courant, P.N., and Porter, R.C. (1981). Averting Expenditure and the Cost of Pollution. Journal of Environmental Economics and Management 8: 321–29.

Cragg, J. G. (1971). Some statistical models for limited dependent variables with application to the demand for durable goods. Econometrica 39: 829–844.

Darley, J. M. (1978). Energy conservation techniques as innovations, and their diffusion. Energy and Buildings 1, 339-343.

De Bartolome C.A.M. (1995). Which Tax Rate Do People Use: Average or Marginal. Jounal of Public Economics 56(1): 79–96.

Deschenes, O., and Greenstone, M. (2011). Climate change, mortality, and adaptation: Evidence from annual fluctuations in weather in the US. American Economic Journal: Applied Economics, 3 (4): 152-185.

Dickie, M. (2017). Averting Behavior Methods. In: Champ P., Boyle K., Brown T. (eds) A Primer on Nonmarket Valuation. The Economics of Non-Market Goods and Resources, vol 13. Springer, Dordrecht

Dickie, M., and Gerking, S. (1991). Willingness to pay for ozone control: Inferences from the demand for medical care. Journal of Environmental Economics and Management, 21: 1-16.

Doria, M.F. (2010). Factors influencing public perception of drinking water quality. Water Policy 12(1): 1-19.

Doria, M.F., Pidgeon, N., and Hunter, P.R. (2009). Perceptions of drinking water quality and risk and its effect on behaviour: A cross-national study. Science of the Total Environment 407 (21): 5455-5464.

Dupont, D., Adamowicz, W.L. and Krupnick, A. (2010). Differences in water consumption choices in Canada: The role of socio-demographics, experiences, and perceptions of health risk. Journal of Water and Health 8(4): 671-686.

European Environment Agency (2012). Towards efficient use of water resources in Europe. Copenhagen: European Environment Agency. Technical report 1/2012.

Estrada, F., Wouter Botzen, W.J. and Tol, R.S.J. (2017). A global economic assessment of city policies to reduce climate change impacts, Nature Climate Change 7: 403–406.

Farbaim, et al. (2016). Sources and transport of contaminants of emerging concern: A two-year study of occurrence and spatiotemporal variation in a mixed land use watershed. Science of the Total environment.

Gerking, S., and Stanley, L. R. (1986). An Economic Analysis of Air Pollution and Health: The Case of St. Louis', Review of Economics and Statistics 68(1): 1151–1121.

Gleick, P. H., and Cooley, H. S. (2009). Energy implications of bottled water. Environmental Research Letters 4(1): 1–6.

Greene, W. (2012). Econometric Analysis. Seventh Edition. Prentice Hall. New Jersey, United States of America.

Harrington, W., Krpnick, A. J., and Spofford, W. A. (1989). The Economic Losses of a Waterborne Disease Outbreak. Journal of Urban Economics 25: 116–137.

Harford, J. (1984). Averting Hevavior and the benefits of reduced soiling. Journal of Environmental Economics and Management 11: 296–302.

Hayes, D. (1976). Energy: The case for conservation (Paper No. 4). Washington, D.C.: Worldwatch Institute.

Heckman, J. (1979). Sample Selection Bias as a Specification Error. Econometrica 47: 153–161.

Hu, Z., Morton, L.W., and Mahler, R.L. (2011). Bottled water: United States consumers and their perceptions of water quality. International Journal of Environmental Research and Public Health 8(2): 565-578.

Instituto Nacional de Estadística (INE) (2015a). Cifras de población Año 2015. Available at http://www.ine.es/inebaseDYN/cp30321/cp_resultados.htm

Instituto Nacional de Estadística (INE) (2015b). Encuesta Continua de Hogares Año 2014. Nota de prensa 903. Available at http://www.ine.es/prensa/np903.pdf

International Bottled Water Association (2015). Water and Energy Use. Benchmarking Study: Executive Summary. Antea Group.

Isaac, M., and van Vuuren D.P. (2009). Modeling global residential sector energy demand for heating and air conditioning in the context of climate change, Energy Policy, 37: 507-521

Jakus, P.M., Shaw, W.D., Nguyen, T.N, and Walker, M. (2009). Risk perceptions of arsenic tap water and consumption of bottled water. Water Resources Research 45(5). DOI: 10.1029/2008WR007427.

Janmaat, J. (2007). A little knowledge…: Household water quality investment in the Annapolis Valley. Canadian Journal of Agricultural Economics 55(2): 233-253.

Johnstone, N., and Serret, Y. (2012). Determinants of bottled and purified water consumption: Results based on an OECD survey. Water Policy 14 (4): 668-679.

Jones, A. (1989). A Double-Hurdle Model of Cigarette Consumption: Summary. Journal of Applied Econometrics 4: 23-39.

Kahneman, D., and Tversky, A. (1982). The Psychology of Preferences. Scientific American 246: 160–173.

Kollmuss, A., and Agyeman, J. (2002). Mind the gap: why do people act environmentally and what are the barriers to pro-environmental behavior? Environmental Education Research 8(3): 239–260.

Larson, B.A., and Gnedenko, E.D. (1999). Avoiding health risks from drinking water in Moscow: An empirical analysis. Environment and Development Economics 4(4): 565-581.

Laughland, A. S., Musser, W. N., Shortle, J.S., and Musser, L. M. (1996). Construct Validity of Averting Cost Measures of Environmental Benefits. Land Economics 72: 100–112.

Lin, T-F., and Schmidt, P. (1984). A test of the Tobit Specification against an alternative Suggested by Cragg. Review of Economics and Statistics 66 (1): 174-77

Lloyd-Smith, P., Schram, C., Adamowicz, W., and Dupont, D. (2015). Endogeneity of Risk Perceptions in Averting Behavior Models. Environmental and Resource Economics. Available online. DOI 10.1007/s10640-016-0075-6

Mansfield, C., Johnson, F. R., and Van Houtven, G. (2006). The missing piece: Valuing averting behavior for children's ozone exposures. Resource and Energy Economics 28: 215-228.

MHAP, 2016. Encuesta de infraestructuras y equipamientos locales 2016. Ministerior de Hacienda y Administraciones Públicas. Madrid. Available at https://ssweb.seap.minhap.es/ (Accessed October 20, 2016).

Murdoch, J. C., and Thayer, M. A. (1990). The benefits of reducing the incidence of nonmelanoma skin cancers: A defensive expenditures approach. Journal of Environmental Economics and Management 18: 107-119.

Nataraj, S., and Hanneman, M. (2011). Does marginal price matter? A regression discontinuity approach to estimating water demand. Journal of Environmental Economics and Management 61(2): 198–212.

Neidell, M. (2009). Information, avoidance behavior and health: The effects of ozone on asthma hospitalizations. Journal of Human Resources 44: 450-478.

Nieswiadomy, M. L., and Molina, D.J. (1989). Comparing residential water estimates under decreasing and increasing block rates using household data. Land Economics 65 (3): 280–289.

Olmstead, S.M., and Stavins, R.N. (2007). Managing Water Demand. Price vs. Non-price conservation programs. A Pioneer Institute Water paper no 39.

Pérez-Urdiales, M., García-Valiñas, M.A., and Martínez-Espiñeira, R. (2015). Responses to changes in domestic water tariff structures: A Latent Class Analysis on household-level data from Granada, Spain. Environmental and Resource Economics 63: 167–191

Salamanca, F., Georgescu M., Mahalov, A., Moustaoui, M., and Wang, M. (2014). Anthropogenic heating of the urban environment due to air conditioning. Journal of Geophysical Research Atmospheres 119 (27): 5949–5965.

Smith, V.K., and Desvouges, W.H. (1986). Averting Behavior: Does It Exist?. Economic Letters 20: 3291–96.

Simon, H.A. (1955). A Behavioral Model of Rational Choice. The Quarterly Journal of Economics 69 (1): 99-118.

Slovic, P. (2000). The Perception of Risk. Earthscan: London

Slovic, P. (1987). Perception of risk. Science, 236: 280-285.

Stern, P.C., and Gardner, G.T. (1981). Psychological research and energy policy. Am. Psychol. 36: 329–342.

Suárez-Varela, M. and Dinar, A. (2017). A Double-hurdle Approach for Estimation of Bottled Water Demand under Consumer Environmental Attitudes and Water Conservation Policies. UCR SPP Working Paper Series, January, 2017 WP# 01-17. (http://spp.ucr.edu/publications/bottled-water-demand.pdf.)

Thomas, M., and Morwitz, V.G. (2009) The Ease-of-Computation Effect: The Interplay of Metacognitive Experiences and Naive Theories in Judgments of Price Differences. Journal of Marketing Research 46 (1): 81-91.

Tobin, J. (1958). Estimation of relationships for limited dependent variables. Econometrica 26: 24–36.

Tovar-Sánchez, A., Sánchez-Quiles, D., Basterretxea, G., Benedé, J.L., Chisvert, A., Salvador, A., et al. (2013). Sunscreen Products as Emerging Pollutants to Coastal Waters. PLoS ONE 8(6): e65451. https://doi.org/10.1371/journal.pone.0065451

Tversky, A., and Kahneman, D. (1973). Availability: A Heuristic for Judging Frequency and Probability. Cognitive Psychology 5 (2): 207–232.

Um, M.J., Kwak, S.J., and Kim, T.Y. (2002). Estimating Willingness to Pay for Improved Drinking Water Quality Using Averting Behavior Method with Perception Measure. Environmental and Resource Economics 21 (3): 287–302.

United Nations (UN) (2015). Progress on Sanitation and Drinking Water. WHO.UNICEF Joint Monitoring Programme for Water Supply and Sanitation (JMP). New York, United States of America.

U.S.EPA. (1998). Pesticide Reregistration Eligibility Decisions (REDS) Database on DEET (134-62-3). USEPA-738/R- 98-010.

Wooldridge, J. (2010). Econometric Analysis of Cross Section and Panel Data. The MIT Press, Cambridge, MA, 2nd edition.

Yoo, S.-H., and Yang, C.-Y. (2000). Dealing with bottled water expenditure data with zero observations. Economics Letters 66: 151–157.

Yoo, S.-H (2003). Application of a mixture model to approximate bottled water consumption distribution, Applied Economics Letters 10 (3): 181-184. DOI:10.1080/1350485022000044101

Zivin, G.J., Neidell, M., and Schlenker, W. (2011). Water Quality Violations and Avoidance Behavior: Evidence from Bottled Water Consumption. American Economic Review: Papers and Proceedings, 101(3).

Essay 4: Ownership and performance in water services revisited: Does private management really outperform public?

Abstract

Since the late 1970s, water services have been privatised in some developed countries in an attempt to improve performance. However, after three decades of privatisations the superiority of private management is being called into question and several cities are returning to public provision. In this paper we revisit the relationship between ownership and performance in urban water services management using directional distance functions, metafrontiers and *Data Envelopment Analysis* (DEA) techniques. The technical efficiency in the provision of water delivery services in a sample of Spanish municipalities is assessed at the level of the management of specific production factors; moreover, we discuss

whether differences in efficiency between private and public decision units are due to either different capabilities of managers (managerial efficiency) or different technological restrictions (ownership efficiency). Our main finding is that private management is more efficient in the use of labour input, mainly because of the technological restrictions faced by public management units, such as legal and institutional restrictions. Conversely, private management appears to be less efficient at managing operational costs.

4.1. Introduction

The issue of the relationship between ownership and performance has long been at the core of the debate in the water industry. Water is a merit good that serves economic, environmental and social purposes (OECD, 2003), and displays important positive externalities. It is also a human need, so universal access should be guaranteed. Moreover, the water industry faces high fixed costs and is very capital intensive with high initial investment required, conditions which lead to a natural monopoly. Accordingly, thorough supervision and intervention on behalf of the public sector is justified as a means of preventing market failure, achieving an efficient allocation of resources and guaranteeing that welfare standards are met (Pigou, 1932). Out of a number of possible options, one of the leading forms of intervention is by means of public companies. In fact, over the years, public provision has been the most common form of water services provision (Thomas *et al.*, 2012).

However, following the wave of deregulation of economic activity that started in Anglo-Saxon economies in the late 1970s, private sector participation in the water industry became increasingly popular and nowadays it is widespread in some developed countries (see Pérard, 2009). Deregulation was based on the idea that, far from pursuing the general interest, public intervention works to satisfy political interests (Niskanen, 1971); accordingly, privatising water provision and introducing competition via tendering processes should promote efficiency and cost reduction. In addition, privatisation would allow the aggregation of demand, particularly in small-sized municipalities, thus achieving a more efficient scale of production (Donahue, 1989). Other strands of thought have highlighted, nonetheless, that the predicted improvements in efficiency resulting from privatisation and competition can be hampered by the existence of transaction costs (Coase, 1937), incomplete information, incomplete contracts or high asset specificity (Williamson, 1976). Also, Donahue's argument that privatisation allows for the aggregation of demand appears to neglect the fact that intermunicipal provision is possible and in fact often does take place under public ownership, e.g., in the form of intermunicipal consortia.

Several papers have found that, although generalisations about the factors that explain privatisation in the urban water service should be drawn carefully (Ruíz-Villaverde *et al.*, 2015), in practice this decision is mainly motivated by pragmatic reasons, including budget restrictions or searching for cost reduction and efficiency gains, rather than ideological or political issues (Bel & Fageda, 2007; 2009). Furthermore, after more than three decades of research, empirical evidence as to the superiority of private management of urban water services over public is inconclusive, and several cities are moving back to public provision. Indeed, a number of municipalities in developed countries, including notable European cities such as Berlin and Paris, have remunicipalised the provision of urban water services in recent years (González-Gómez *et al.*, 2009; Hall *et al.*, 2013), while growing opposition to new privatisations is emerging from citizens' movements and certain political parties (Hall *et al.*, 2005; Lobina *et al.*, 2014). Pigeon *et al.* (2012) analysed a series of case studies of water services remucipalisation from a comparative international perspective, concluding that back-to-public provision largely occurred in response to the failure of the preceding privatisation.

On the other hand, as we have mentioned above and is detailed in Section 4.2 below, an extensive literature has focused on a comparative assessment of the operational performance of public and private operators. However, the empirical evidence is not at all conclusive. In these papers, researchers have overwhelmingly tended to assume that both categories of management –public and private– share the same production technology. To our knowledge, there are only a few cases that consider the possibility that public and private managers could face different technological restrictions (see Mbuvi, 2012). However, as we argue in this paper, there are reasons to believe that the production technology might differ according to the nature of operator ownership; additionally, these differences might well have disparate effects on the management of particular production factors.

Against this background, this paper revisits the relationship between ownership and performance in water utilities using a fresh methodological approach that combines directional distance functions, metafrontiers and *Data Envelopment Analysis* (DEA) techniques. Specifically, the performance of a sample of both public and private operators in the Spanish urban water industry is assessed through the concept of *technical efficiency*, understood as their ability to reduce input usage for a given volume of output.

Our contribution to existing literature in this field is twofold. First, using metafrontiers allows us to express *technical efficiency* as the result of *managerial efficiency*, which assesses the performance of operators in the sample as compared to best practices in their group, either public or private operators, and *ownership efficiency* that measures the closeness of the technology of each group to the joint technology. In this way, differences in technical efficiency between private and public operators can be attributed to either the different capabilities of their managers (managerial efficiency), or the different technological restrictions (i.e., legal and/or institutional restrictions) faced by these two types of management (ownership efficiency). Second, and more interestingly, directional distance functions allow us to evaluate performance at the level of the management of specific production factors, including labour and other operational costs. In our opinion, there are reasonable grounds to believe that the technological constraints faced by public managers may differ from those faced by their private counterparts in terms of managing labour, which would not affect the management of other production factors. The combination of these two approaches has the potential to provide new insights into the relationship between ownership and performance in the water industry, as opposed to previous papers in which either metafrontiers or directional distance functions have been used separately.

In our empirical analysis, we use information about the provision of water delivery services for a sample of Spanish municipalities. Our main finding is that private management is more efficient in the use of labour, mainly because of technological restrictions faced by public operators when managing this input. Conversely, private operators appear to be less efficient at managing operational costs, although this result is statistically less robust.

The remainder of the paper is organised as follows. Section 4.2 briefly reviews the empirical literature on ownership and performance in the provision of water services. Section 4.3 presents the data and explains the methodology. Section 4.4 describes and discusses the results, while the final section summarises and concludes.

4.2. Ownership and performance in water utilities: Some empirical evidence

Research on the effects of privatisation on the efficiency of water services management dates back to the 1970s, with the seminal works by Mann & Mikesell (1976), Morgan (1977) and Crain & Zardkoohi (1978). Those first studies focused on the water industry in the Unites States and, since then, this issue has been the subject of increasing attention. By the end of the eighties only around thirty papers had been published, whereas in the nineties alone about forty studies were conducted, and by 2010 there were well over two hundred and fifty publications (Berg & Marques, 2011; Carvalho *et al.*, 2012). Moreover, the initial geographical focus quickly spread to areas with markedly different contexts, so that case studies can now be found from the five continents, and from both developing and developed economies.

Bel & Warner (2008) and Bel *et al.* (2010) carried out two meta-regression analyses of empirical studies finding little support for a link between privatisation and cost savings in solid waste and water services; in particular, cost savings are not found in water delivery, while they are not systematic in waste. Similarly, Lobina (2013) critically reviewed empirical literature on organisational choice and efficiency in the urban water sector, suggesting that institutional adaptability explains the efficiency and effectiveness of the public sector relative to the private sector. For the purpose of our review, empirical studies addressing the issue of differences in efficiency between publicly and privately managed urban water services will be classified into two broad groups: efficiency assessments for the same operator(s) in function of changes in ownership (through time); and efficiency assessments comparing operators under different ownership regimes (at given points of time).

Most studies in the first of the aforementioned groups focus on the massive privatisation of the water industry witnessed in the United Kingdom at the end of the 1980s, and they are in general agreement that privatisation did not lead to increased efficiency in urban water services. By way of example, Saal & Parker (2000) found no evidence supporting a relationship between privatisation and efficiency improvements in the water and sewerage industry in England and Wales. Likewise, Saal & Parker (2001) suggested that despite reduction in labour usage, total factor productivity growth in water and sewage companies did not improve following the privatisation of the industry; conversely, utilities' economic profits increased. Other papers in this line are Ashton (2000), Saal & Parker (2004) and

Saal *et al.* (2007).[70] An exception is the paper by Estache & Trujillo (2003) that, using information from four utilities in Argentina between 1992 and 2001, found that privatisation led to important gains in total factor productivity. However, as pointed out by the authors, this result should be interpreted with caution given the small size of the sample employed.

Conclusions from the second group of studies are more diverse and even contradictory. Without aiming to be exhaustive, Table 31 shows a selection of empirical studies. Among those papers that find public management of urban water services to be superior, several explanations are adduced, such as lower costs (Mann & Mikesell, 1976; Bruggink, 1982; Bhattacharyya *et al.*, 1995a) or better results in a range of performance indicators (Chong *et al.*, 2006; Benito *et al.*, 2010; Romano & Guerrini, 2011; Guerrini, 2011; Da Cruz *et al.*, 2012; Romano *et al.*, 2013; Lannier & Porcher, 2014). Other studies find that public companies are also more efficient at achieving social and development goals (Lobina & Hall, 2000). Regarding analyses that found private management to be superior in terms of performance, reasons given also include lower costs (Morgan, 1977; Crain & Zardkoohi, 1978) and higher technical efficiency (Picazo-Tadeo *et al.*, 2009a, 2009b). In addition, some of these papers maintain that differences in efficiency are mainly related to labour management (Crain & Zardkoohi, 1978; Picazo-Tadeo *et al.*, 2009a, 2009b; Gassner *et al.*, 2009). Nevertheless, a majority of studies either find no significant difference between the performance of public and private water suppliers or reach no definite conclusion. Furthermore, some papers point out that once characteristics of the operating environment are accounted for, differences in efficiency diminish (Ménard & Saussier, 2000; González-Gómez *et al.*, 2013).

[70] In addition, some papers have analysed the impact of changes in regulation on the performance of the privatised English and Welsh water industry (Erbetta & Cave, 2007; Maziotis *et al.*, 2016).

Table 31: Public versus private management of water services: Some empirical studies.

Superiority of public management	*Superiority of private management*	*No significant difference or inconclusive*
Mann & Mikesell (1976)	Morgan (1977)	Feigenbaum & Teeples (1983)
Bruggink (1982)	Crain & Zardkoohi (1978)	Byrnes *et al.* (1986)
Lambert *et al.* (1993)	Bhattacharyya *et al.* (1995b)	Ménard & Saussier (2000)
Bhattacharyya *et al.* (1994)	Estache & Kouassi (2002)	Estache & Rossi (2002)
Bhattacharyya *et al.* (1995a)	Faria *et al.* (2005)	Kirkpatrick *et al.* (2006)
Lobina & Hall (2000)	Picazo-Tadeo *et al.* (2009a;b)	García-Sánchez (2006)
Benito *et al.* (2010)	Lo Storto (2013)	Sabbioni (2008)
Romano & Guerrini (2011)		Zschille & Walter (2012)
Guerrini *et al.* (2011)		Peda *et al.* (2013)
Da Cruz *et al.* (2012)		González-Gómez *et al.* (2013)
Romano *et al.* (2013)		Hon *et al.* (2014)
Lannier & Porcher (2014)		

Regarding the methodological approach employed to assess efficiency, until the beginning of the 21st century there was a predominance of parametric techniques using cost (Mann & Mikesell, 1976; Morgan, 1977; Bruggink, 1982; Feigenbaum & Teeples, 1983; Bhattacharyya *et al.*, 1994) and production (Crain & Zardkoohi, 1978) functions, and/or *Stochastic Frontier Analysis* (SFA) (see Aigner *et al.*, 1977, and Meeusen & van den Broeck, 1977 for details). However, most studies these days are based on estimates of non-parametric frontiers and performance indicators by means of *Data Envelopment Analysis* (DEA) techniques, with only a few studies using other techniques (Byrnes *et al.*, 1986; Saal & Parker, 2001; Lobina & Hall, 2000; Estache & Trujillo, 2003). DEA is a well-known non-parametric approach to efficiency measurement based on mathematical programming pioneered by Charnes *et al.* (1978) that has been used in hundreds of empirical papers (Cook & Seiford, 2009 and Liu *et al.*, 2013 review this literature). This technique provides a simple way to measure the gap that separates individual producers' behaviour from best productive practices, which are assessed from actual observations of efficient producers' production processes. DEA offers an important advantage over the econometric approach to efficiency measurement, as it allows the technological frontier representing best-observed practices to be flexibly constructed without imposing a given functional form on either technology (e.g., the Cobb-Douglas or the translog production functions) or inefficiencies (e.g., distribution functions such as the half-normal). More details on DEA techniques can be found in Cooper *et al.* (2007).

On the other hand, performance differences between public and private management units have been evaluated using two main methodological approaches. The first consists of using conventional *ANOVA*, *Mann-Whitney*, *Kolmogorov-Smirnov* or *Kruskal-Wallis* tests, among others, to test for differences in efficiency scores obtained from either DEA-based analyses or cost and production function estimates with SFA. The second approach relies on directly including dummy variables reflecting ownership in the estimation of cost and production functions with SFA, or including them in second- or third-step regression analyses of DEA-based efficiency scores.

In summary, we believe that the lack of conclusive evidence in previous literature calls for fresh methodological and empirical approaches to assessing the relationship between efficiency and ownership in the water industry, and consequently we attempt such an approach in this paper.

4.3. Data, variables and methodology

4.3.1. Data and sample

In this paper we use data relating to the provision of urban water delivery services[71] in 70 Spanish municipalities of under 50,000 inhabitants. In 37 of these municipalities either the city council itself or a public utility manages water delivery

[71] In addition to water delivery, some water utilities in Spain also provide sewage treatment services; however, this is not the case with the operators in our sample.

(public management units), while in the other 33 cases the service is privately managed by either a contractual public-private partnership (PPP) or an institutionalised PPP (private management units).[72] The data are from 2013 and were collected, when available, from web pages of municipalities and utilities as well as by direct contact with city councils and utilities' managers, in the framework of a wider project supported and financed by the Spanish *Ministry of Economy and Competitiveness*.[73]

Two outputs and three inputs are used to characterise the productive process of both public and private operators. The two outputs are water delivered and population served. Of the three inputs, one is fixed –the length of the delivery network–, and two are variable –labour (full-time workers) and operational costs (measured in euros)[74] (see Picazo-Tadeo *et al.*, 2008). Table 32 provides measurement units and some descriptive statistics for the data.

[72] García-Valiñas *et al.* (2013) provides a detailed description of legal forms for the management of urban water services in Spain. Furthermore, following previous literature, institutionalised PPPs have been considered as private management units given that day-to-day management is carried out by the private partner (see García-Valiñas *et al.*, 2013; Picazo-Tadeo *et al.*, 2012). Finally, it is worth mentioning that in compliance with Spanish law only the management of the urban water service can be privatised, while infrastructures always remain under public property.

[73] Spanish legislation prevents data on inputs and outputs of water suppliers from being made public. When creating our database, we submitted information requests to nearly 1,000 Spanish municipalities, either via web pages or directly to city councils and utilities. Of these, we received 141 positive responses. After discarding observations with deficient or incomplete information, we selected 70 operators that are exclusively dedicated to water service delivery. Unfortunately, the aforementioned lack of publicly-available information makes it very difficult to obtain reliable and largely representative data on the production processes of Spanish water services operators. This is reflected in previous studies on Spanish water utilities, which make use of samples of similar size (Picazo-Tadeo *et al.*, 2011; Picazo-Tadeo *et al.*, 2009a; 2009b).

[74] Operational costs include all expenses required for day-to-day management of the service, e.g., raw water, chemicals employed to make water suitable for human consumption, energy and office expenses, among others. Conversely, wages and other labour costs are excluded. Furthermore, the fee paid by utilities to the local government when they are first awarded the service management contract is also excluded from operational costs. Finally, it is worth highlighting that operational costs are measured in euros, which means that computed technical efficiency might also include a component of allocative (price) efficiency. This is, however, a common problem in efficiency analyses that would have only a minor impact on the measurement of technical efficiency if production factor markets are assumed to be competitive with small price differences.

Table 32: Sample descriptive statistics.

	Measurement unit	Mean	Standard deviation	Maximum	Minimum
Public management					
Water delivered	*Thousands of m^3*	510.4	526,6	2,365.3	55.0
Population served	*Thousands*	7.4	7.2	25.3	1,1
Labour	*Full-time workers*	4.5	2.7	11.0	0.5
Operational costs	*Thousands of €*	415.5	532.2	2,351.9	44.5
Distribution network	*Kilometres*	43.6	33.6	163.7	7.7
Private management					
Water delivered	*Thousands of m^3*	1,561.0	1,120,0	3,475.5	113.4
Population served	*Thousands*	21.2	13.9	43.8	1.6
Labour	*Full-time workers*	12.6	10.5	43.0	1.6
Operational costs	*Thousands of €*	1,414.9	986.9	3,767.9	74.1
Distribution network	*Kilometres*	157.2	230.0	1,330.8	24.1

4.4. Methodological issues

4.4.1. The metatechnology and the group technology

Our methodological approach is based on Sáez-Fernandez *et al.* (2012), which uses directional distance functions (Chambers *et al.*, 1998) to extend the metafrontier approach by O'Donnell *et al.* (2008) to the measurement of technological differences in the management of specific inputs.[75] In order to develop the main insights of this approach, let us assume that our k = 1…70 decision units (operators) use the set of inputs $x = (x_f, x_v)$, where the fixed input x_f is the length of the delivery network, and variable inputs x_v are labour and operational costs, to produce the vector of outputs y, which includes water delivered and population served.

[75] See Beltrán-Esteve (2013), Beltrán-Esteve *et al.* (2014) and Picazo-Tadeo *et al.* (2014) for recent empirical applications of this approach.

Transformation of inputs into outputs requires the use of a *metatechnology* that is represented by the *short-run input requirement set*. This set includes all combinations of variable inputs x_v that, given a fixed input endowment x_f, allow production of at least a level of outputs y. It is formally defined as:

$$L\ (x_f, y\ )\ \equiv\ [x_v \mid (x, y) \in T] \tag{23}$$

where T represents all combinations of inputs and outputs attainable with the present state of knowledge. It is assumed that the metatechnology satisfies the standard properties suggested by Shephard (1970).

The instrument used to compare the production plan of each decision-making unit in our sample with respect to best available practices[76] in the metatechnology, i.e., the technological frontier, is the *directional metadistance function* defined as (Färe & Grosskopf, 2000):

$$\begin{aligned} M\overrightarrow{D} = \ & \left[x, y;\ \ g = \left(-g_{x_v},\ g_y\right)\right] = \\ & = Sup\ \left\langle \delta \mid \left(x_v -\ \delta g_{x_v}\right) \epsilon\ L\ \left[x_f, \left(y + \delta g_y\ \right)\right] \right\rangle \end{aligned} \tag{24}$$

With $g\ = (-g_x, g_y)$ being the so-called direction vector.

This function has a lower bound of zero (other properties are in Chambers *et al.*, 1998), and models inputs and outputs jointly by seeking the maximum attainable expansion of outputs in the g_y direction and the largest feasible contraction of variable inputs in the $-g_{xv}$ direction. Furthermore, the directional metadistance function is a very flexible tool for assessing efficiency as it allows the technological frontier to be approached via alternative paths which focus on different facets of performance (Picazo-Tadeo *et al.*, 2012).

These paths might represent the preferences of utilities' managers and/or policymakers regarding performance. If we were interested in assessing the maximum proportional (radial) feasible reduction of variable inputs labour and

[76] In this general setting, best practices are determined by those productive plans, either observed productive plans or resulting from their linear combinations, which obtain more outputs with fewer variable inputs usage, always for given endowment of the fixed input.

operational costs, given the endowment of the fixed input delivery network and also maintaining the level of outputs, the directional metadistance function would be:

$$\begin{aligned} M\vec{D}_{radial} = \ & [x, y; \ \ g_{radial} = (-x_v, 0)] = \\ & = Sup \ \langle \delta_{radial} \mid (1 - \ \delta_{radial}) \ x_v \ \epsilon \ L \ (x_f, y) \rangle \end{aligned} \tag{25}$$

By way of example, a score of 0.1 for a particular operator in our sample would mean that, given the length of its delivery network, it could reduce both labour and operational costs by 10% without any decrease in the amount of water delivered or the population served.

Furthermore, it might be of interest to assess the potential reduction of variable input i, either labour or operational costs, while maintaining the other input –i, always for given fixed input and outputs, i.e., assessing technical efficiency in the management of variable input i. In this scenario, the directional metadistance function becomes:

$$\begin{aligned} M\vec{D}_i = \ & \langle x, y; \ \ g_i = [(-x_{v_i}, 0_{v_{-1}}), 0] \rangle \\ & = Sup \ \langle \delta_i \mid [(1 - \ \delta_i) \ x_{v_i}, \ x_{v_{-i}} \ ] \ \epsilon \ L \ (x_f, y) \rangle \end{aligned} \tag{26}$$

In this case, a score of, say, 0.2 for the directional metadistance function and labour input would indicate that the number of full-time workers could be reduced by 20% without increasing operational costs and, importantly, while still maintaining the amount of water delivered and population served.

The directional distance functions of expressions (3) and (4) can also be computed with respect to the technology of the two groups of operators considered in this research, namely, public and private. Accordingly, the technology of group h (with h = public, private) is based only on observations of decision units within this group, and can also be represented by the *short-run input requirement set* defined as:

$$L^h \ (x_f, y \ ) \ \equiv \ [x_v \mid (x, y) \in T^h] \tag{27}$$

with T^h representing all the combinations of inputs and outputs attainable by operators in group h, i.e., the state of knowledge for units in that group. The key issue here is that some productive plans, i.e., combinations of inputs and outputs, included in the metatechnology may not be possible given the technology of a particular group.

Formally, the directional distance functions computed with respect to the technology of group h in the case of radial and specific reduction of inputs are, respectively:[77]

$$\begin{aligned} \vec{D}^{h}_{radial} = \ & [x, y;\ \ g_{radial} = (-x_v, 0)] \\ & = Sup\ [\delta^{h}_{radial} \mid (1 - \ \delta^{h}_{radial})\ x_v\ \epsilon\ L^{h}\ (x_f, y)] \end{aligned} \tag{28}$$

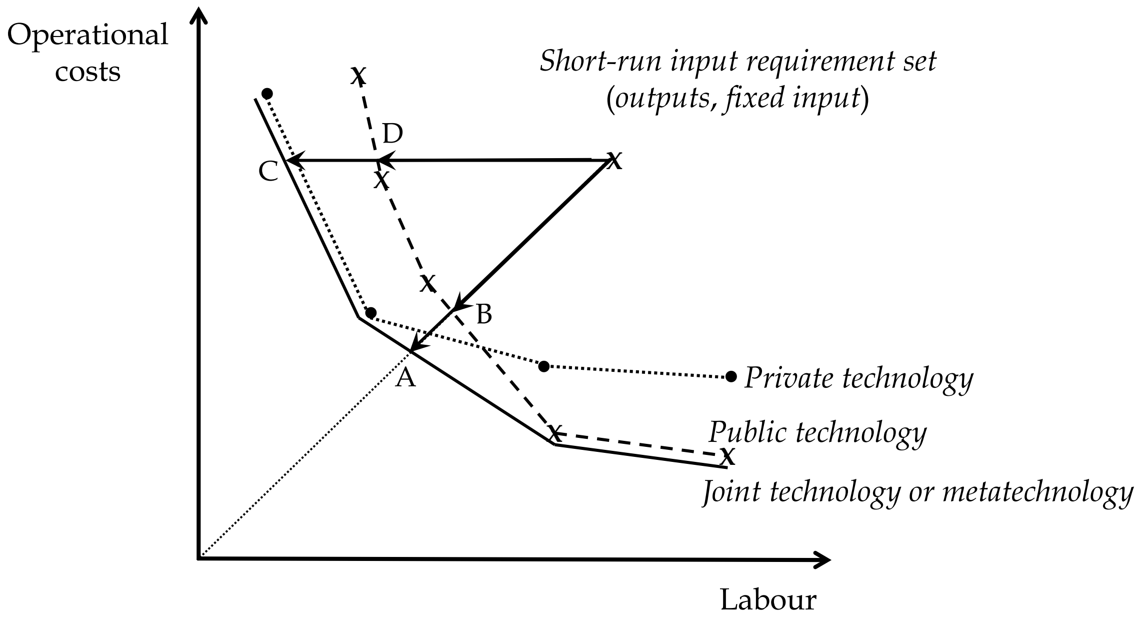
and,

$$\begin{aligned} \vec{D}_i^{\ h} = \ & \langle x, y;\ \ g_i = \left[(-x_{v_i}, 0_{v_{-1}}), 0\right] \rangle \\ & = Sup\ \langle \delta_i^{\ h} \mid \left[(1 - \ \delta_i^{\ h})\ x_{v_i},\ x_{v_{-i}}\ \right]\ \epsilon\ L^{h}\ (x_f, y) \rangle \end{aligned} \tag{29}$$

Figure 6 provides a graphical illustration of our directional functions. For the sake of simplicity, this is a hypothetical scenario in which we observe a set of four private management units represented by dots, and another set of six public management units identified by crosses. The short-run metatechnology or joint technology is represented by the lower envelope of all these observations regardless of their private or public character, i.e., the isoquant represented by the continuous line. Similarly, the technologies of private and public units are represented by the dotted and dashed isoquants, respectively. Projecting the inefficient public operator, i.e., the one located in the interior of the input requirement set, onto the metatechnology with a direction that reduces both labour and operational costs simultaneously yields point A; furthermore, projection onto the technology of the group of public units would yield point B. Accordingly, the segment BA measures the distance that separates the public technology from the metatechnology evaluated at this projection, i.e., the metatechnology ratio. Similarly, the segment DC measures the distance that separates the public technology from the metatechnology, assessed in a direction that reduces labour input whilst operational costs are maintained.

[77] By construction, directional distance functions computed relative to the technology of group h will always be equal to or lower than directional metadistance functions computed with respect to the metatechnology.

Figure 6: Metatechnology, group technologies and metatechnology ratios.



Source: authors

4.4.2. Computation of the metatechnology ratios

Using the directional distance/metadistance functions defined in Section 4.3.2.1, we have computed the metatechnology ratios proposed by O'Donnell *et al.* (2008). In particular, the *metatechnology ratio* of group h can be defined as:

$$\begin{aligned} Metatechnology\ ratio\ {}^{h}_{radial} = \ [x, y;\ \ g_{radial} = (-x_v, 0)] \\ = \frac{Technical\ efficiency_{radial}}{Technical\ efficiency^{h}_{radial}} = \frac{(1-\ \delta_{radial})}{(1-\ \delta^{h}_{radial})} \end{aligned} \tag{30}$$

It is worth noting that the metatechnology ratio has been defined using technical efficiency scores with an upper bound of one –a score that indicates full efficiency– and can be interpreted as conventional Farrell-type technical efficiency measures (Farrell, 1957).[78,79] Furthermore, technical efficiency scores computed with respect to the technology of group h will be equal to or higher than those computed relative to the metatechnology.

The metatechnology ratio provides a measure of how close the technology of group h is to the metatechnology, assessed in a direction that reduces all variable inputs proportionally. For example, a metatechnology ratio of 0.85 means that the efficient level of variable inputs usage needed to produce a given level of outputs relative to the joint technology is 85% of the efficient usage relative to the technology of the group h, i.e., either public or private management units. According to O'Donnell *et al.* (2008, p. 237), this approach provides a suitable decomposition of technical efficiency assessed with respect to the metafrontier (representing the existing state of knowledge), into the product of technical efficiency measured with respect to the frontier of group h (that represents the state of knowledge as well as physical, regulatory and other restrictions faced by units in that group) and the metatechnology ratio for group h (which measures how close the technology of this group is to the joint technology). Formally:

$$\begin{aligned} &Technical\ efficiency_{radial} \\ &\quad = Technical\ efficiency^{h}_{radial} \cdot Metatechnology\ ratio\ ^{h}_{radial} \end{aligned} \tag{31}$$

In less technical terms, this approach allows the decomposition of *technical efficiency* into *managerial efficiency*, which assesses the performance of operators in the sample as compared to best practices in their group, and *ownership efficiency*, which measures the closeness of the technology of group h to the joint technology.

[78] The reason for this choice is that, although directional metadistance/distance functions can also be directly interpreted as measures of technical efficiency, distances for efficient management units are equal to zero and, thus, metatechnology ratios would not be defined for these operators (Sáez-Fernández *et al.*, 2012).

[79] For example, a score for the directional distance function in the radial scenario of 0.1 would indicate, as already mentioned, that outputs could be maintained while reducing labour and operational costs by 10%. In this case, the technical efficiency score would be 0.9, indicating that it would be possible to maintain the same level of water delivered and population served with only 90% of observed inputs usage.

Similarly, the *input-specific metatechnology ratio* for variable input i and group h is:

$$\begin{aligned} Metatechnology\ ratio\ _i^h = &\ \left\langle x, y;\ \ g_i = \left[\left(-x_{v_i}, 0\right), 0\right]\right\rangle \\ = &\ \frac{Technical\ efficiency_i}{Technical\ efficiency_i^h} = \frac{(1 -\ \delta_i)}{(1 -\ \delta_i^h)} \end{aligned} \quad (32)$$

The interpretation of this metatechnology ratio is analogous to that in expression (8), with the difference that now the closeness of group h's technology to the metafrontier is assessed in a direction that only reduces input i without increasing the usage of input –i and maintaining outputs. The abovementioned decomposition of technical efficiency also holds.

Lastly, the directional metadistance/distance functions involved in our analysis have been computed with *Data Envelopment Analysis* (DEA) techniques, using the programs detailed in the *Appendix*.

4.5. Results and discussion

In the conventional scenario that assesses potential proportional reductions of both variable inputs given the fixed input endowment and, also, maintaining outputs, the average for radial efficiency calculated with respect to the metatechnology or joint technology is 0.568 (Table 33).[80] This score suggests that when all operators in our sample are compared with best available practices, labour and operational costs could both be proportionally reduced by an average of 43.2%.[81] For public and private units considered separately, averages of radial technical efficiency are 0.576 and 0.561, respectively, and the difference is not

[80] Note that the exactness of the decomposition of technical efficiency presented in this table does not hold at the aggregate level due to the use of arithmetic means.

[81] This does not necessarily mean that all inefficient operators could adopt the best practices irrespective of the local context in which they develop their productive activity, or without undermining variables such as quality or sustainability. In this sense, research in this field has highlighted how the characteristics of operating environments can affect the technical efficiency of water utilities (Picazo-Tadeo *et al.*, 2009a; 2009b; Ménard & Saussier, 2000; González-Gómez *et al.*, 2013); likewise, service quality also matters in measuring the performance of water utilities (Picazo-Tadeo *et al.*, 2008).

statistically significant at standard confidence levels, according to the results from the *Kolmogorov-Smirnov* and *Mann-Whitney* non-parametric tests (Conover, 1999), and the *Simar-Zelenyuk-Li* test (Simar & Zelenyuk, 2006; Li, 1996). As for managerial efficiency scores, i.e., those computed with respect to the group technologies, averages are 0.669 and 0.682 for public and private operators, respectively. However, it is important to point out that these scores are not directly comparable to each other because they have been obtained with respect to different technological frontiers, and it is well known that efficiency is a relative concept (Färe *et al.*, 1994). Lastly, averages for the metatechnology ratios of public and private units are 0.833[82] and 0.838, respectively, and they are not statistically different according to the results from the tests included in Table 34. The *Kernel* density function represented in Figure 7 provides a graphical illustration of this finding.[83]

Table 33: Estimates of radial technical efficiency.

	Mean	*Standard deviation*
Technical efficiency with respect to the metafrontier	*0.568*	*0.282*
Public management	0.576	0.317
Private management	0.561	0.251
Technical efficiency with respect to the group frontier (managerial efficiency)		
Public management	0.669	0.229
Private management	0.682	0.312
Metatechnology ratio (ownership efficiency)		
Public management	0.833	0.185
Private management	0.838	0.199

[82] This means, by way of example, that the efficient level of labour input usage needed to produce a given output vector relative to the joint technology is 83.3% of the efficient usage relative to the technology of the group of privately managed units.

[83] Table 34 and Figure 7 also include results and *Kernel* density functions obtained in the scenarios of input-specific performance assessment, which are discussed later; *Kernels* have been drawn directly using the metatechnology ratios obtained from expressions (8) and (10).

Table 34: Differences in the metatechnology ratio: Public versus private management.

	Kolmogorov-Smirnov test [a]	*Mann-Whitney test* [b]	*Simar-Zelenyuk-Li test* [c]
	KS-statistic (p-value) [d]	*Z-statistic (p-value)* [e]	*Li-statistic (p-value)* [f]
Radial technical efficiency	0.117 (0.937)	0.445 (0.656)	-0.909 (0.818)
Input-specific technical efficiency			
Labour	0.346 (0.021)**	2.442 (0.014)**	2.989 (0.001)***
Operational costs	0.322 (0.039)**	-2.289 (0.022)**	0.729 (0.232)

*** and **** *stands for statistical significance at 5 and 1 per cent, respectively.*

[a] *The null hypothesis is that the distribution of the two samples is the same.*

[b] *The null hypothesis is that the two samples are drawn from the same population.*

[c] *The null hypothesis is that the two samples have the same probability distribution function.*

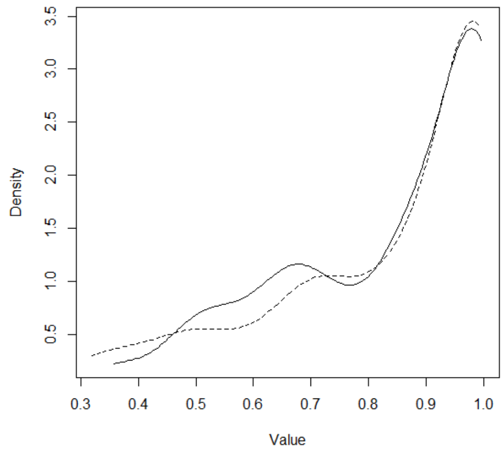
[d] *Exact p-values are provided.*

[e] *Statistics are adjusted for ties.*

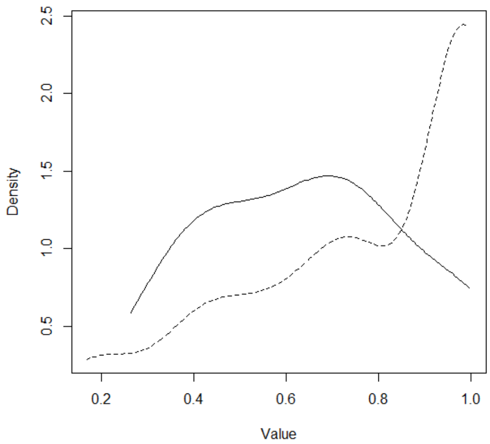
[f] *Original estimates of the metatechnology ratio have been smoothed using Algorithm II in Simar and Zelenyuk (2006).*

Figure 7: Kernel density estimation functions of metatechnology ratios: public (continuous line) versus private (dashed line) management.

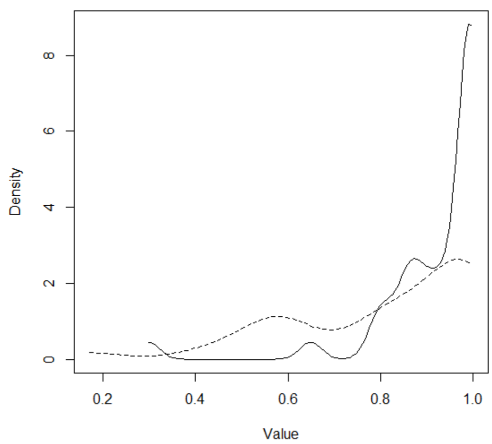
Radial



Labour specific



Operational costs specific



The abovementioned results are in line with most studies in this field, and suggest that there is no significant difference in technical efficiency between public and private management. Nonetheless, the picture is rather different when performance is evaluated at the level of the management of specific production factors, i.e., non-radial measures, which reinforces the relevance of our approach.

In the scenario where only labour input is reduced, technical efficiency averages computed with respect to the metatechnology are 0.402 and 0.480 for public and private decision units, respectively (Table 34); moreover, the difference is statistically significant pointing to the higher efficiency of private management. But what are the reasons for the better performance of private units at managing labour? On the one hand, managerial efficiency scores for public and private units are 0.598 and 0.613, respectively. Although, as mentioned above, these scores are not directly comparable to each other, private units seem to be slightly closer to their technological frontier, on average, than public ones are to theirs. On the other hand, and more interestingly, the metatechnology ratios for public and private units average 0.651 and 0.778, respectively, i.e., the technology of private management units is closer to the metatechnology than the technology of public units is. Moreover, the difference is statistically significant (see Table 35; see also the *Kernel* density functions in Figure 7). In less technical terms, the technology of private operators appears to be more efficient in the management of labour input.[84] This result is in line with Gassner *et al.* (2009), which examined the impact of private sector participation in water distribution in more than 70 developing and transition economies. One of the main findings of that paper is that private participation is associated with gains in performance and labour productivity, which are linked to a reduction in staff numbers. Furthermore, the authors find that private sector also fares better than the public sector in terms of price efficiency. However, efficiency gains under private management are not followed by reduced prices or increased investments, suggesting that '*…the private operator reaps all the gains through profits*' (Gassner *et al.*, 2009, p. 5). Accordingly, efficiency gains from privatisation would not benefit citizens through lower water prices and/or better service quality linked to increased investments, but just to private operators through higher returns.

[84] Picazo-Tadeo *et al.* (2009b) also used a methodological approach based on the computation of input-specific scores of technical efficiency to provide evidence of the superiority of private utilities regarding the management of labour. However, here we go one step further by decomposing technical efficiency into managerial efficiency and ownership efficiency.

Table 35: Estimates of input-specific technical efficiency.

	Labour		Operational costs	
	Mean	*Standard deviation*	*Mean*	*Standard deviation*
Technical efficiency with respect to the metafrontier	*0.439*	*0.325*	*0.499*	*0.325*
Public management	0.402	0.292	0.493	0.294
Private management	0.480	0.358	0.505	0.361
Technical efficiency with respect to the group frontier (managerial efficiency)				
Public management	0.598	0.264	0.539	0.305
Private management	0.613	0.358	0.620	0.373
Metatechnology ratio (ownership efficiency)				
Public management	0.651	0.220	0.920	0.134
Private management	0.778	0.259	0.806	0.214

The superiority of the technology used by private units in the management of labour might be due to certain regulatory and institutional restrictions faced by public management units that could reduce their flexibility in adjusting this production factor. In general, public managers are constrained by more stringent labour regulation which makes it more difficult to fire employees, and they also face higher levels of absenteeism (Meier & O'Toole, 2011). In addition, local governments, particularly those ruled by left-wing parties, tend to develop policies to promote employment stability (Botero *et al.*, 2004; Emmenegger, 2011), as they consider the political costs of cutting jobs to be extremely high. Furthermore, public workers could also emerge as a lobby with a high negotiating power. Finally, creating overemployment when public services are delivered in-house might also form part of local politicians' *rent-seeking* strategy (Hart *et al.*, 1997). Nevertheless, our findings do not allow us to establish a direct causal relationship between these conjectural factors and the superiority of the private technology in managing labour. Nor are a number of further related issues addressed in our research, such as the potential impact of reducing labour usage on the quality of the service offered by private operators. It might, for example, be the case that public operators employ more full-time workers simply because they are essential to delivering higher service quality, a variable that is omitted in our analysis.[85]

Regarding the scenario where only operational costs are reduced, technical efficiency computed with respect to the metatechnology averages 0.493 and 0.505 for public and private operators, respectively (see also Table 34); however, the difference is not statistically significant at the standard confidence levels. This outcome is, nonetheless, the consequence of two contrasting factors. On the one hand, private managers are operating, on average, closer to their own technological frontier than their public counterparts –average scores of managerial efficiency for public and private operators are 0.539 and 0.620, respectively. On the other hand, however, private technology is found to be less efficient at managing operational costs than the technology of public management units: metatechnology ratios for public and private decision units are 0.920 and 0.806, respectively, with the difference being statistically significant according to the *Kolmogorov-Smirnov* and *Mann-Whitney* tests but not the *Simar-Zelenyuk-Li* test (see Table 35; also see Figure 7).

This latter result is then less robust than that obtained in the case of the specific management of labour input and should thus be interpreted with caution. However, several factors could go some way to explaining this. In the first place, cost-sharing activities may take place, especially under in-house public provision. In other words, some operational costs such as administrative costs or energy consumption could be included in the budget item for general municipality expenses, and it would be very difficult to get accurate estimates of the share corresponding to water services. Secondly, it has been shown that there is a tendency to privatise water services operating in more complex environments (González-Gómez *et al.*, 2011), which might imply higher operational costs. For example, some factors that could have an impact on operational costs include the

[85] There is no consensus about the effect of privatisation on the quality of the urban water service, either. In this respect, Galiani *et al.* (2005) found that the privatisation of local water companies in Argentina lead to a significant reduction in child mortality from causes directly related to water conditions such as infectious and parasitic diseases; also Marin (2009) suggested that privatisation in developing countries leads to improved service quality, especially by reducing water rationing. Conversely, Barrera-Osorio *et al.* (2009) showed that privatization in Colombia has strong negative effects on the access to water in rural areas. Furthermore, some papers suggest that privatisation has been followed by deterioration in service quality in the United Kingdom in such aspects as supply continuity and leakage control (Lobina & Hall, 2000; Lobina & Hall, 2001).

state of conservation of the delivery network, the source of raw water and its quality, and network efficiency.[86]

4.6. Conclusions and suggestions for further research

Since the 1980s, a number of papers have studied the relationship between ownership and urban water operators' performance, using a range of conceptual and methodological approaches. Nevertheless, after more than three decades of research, empirical evidence is still inconclusive. Our main contribution to this literature is the use of a fresh approach to assess technical efficiency in the management of water delivery, based on the use of directional distance functions, metafrontiers and *Data Envelopment Analysis*. Unlike extant research, which has used either metafrontiers or directional distance functions separately, combining the two approaches allows us to account for the possibility that public and private operators face different technological restrictions affecting the management of particular production factors. The main advantage of this combined approach is that it allows us to distinguish between managerial efficiency and ownership efficiency at the level of the management of specific inputs, with the latter representing the effect on performance of technological restrictions faced by either public or private ownership regimes.

Furthermore, we focus our empirical analysis on urban water service provision in Spain. Regarding our results, in a conventional scenario based on assessing radial efficiency, as is the case with most previous research, we find no differences of performance between public and private operators; however, when performance is evaluated at the level of the management of specific production factors the picture is somewhat different. On the one hand, the technology of private operators is found to be more efficient in the management of labour, which might be due to certain institutional, regulatory and also political restrictions faced by public management units. Conversely, private operators' technology appears to be less efficient in the management of operational costs, perhaps because they operate in more complex

[86] These hypotheses would need, however, to be empirically tested. Using an indirect approach, we have found that private management is positively correlated with certain variables representing the complexity of operating environments, e.g., a dummy variable that characterises municipalities where intensive treatment is required to make raw water suitable for drinking, and an index of delivery network density computed as kilometres of network per 1,000 inhabitants.

environments, which probably leads to higher operational costs. However, this latter outcome is statistically less robust. In summary, our approach seems to be successful in uncovering new insights hidden to more conventional approaches based on the simple calculation of radial or overall measures of performance in the provision of the urban water service (some exceptions are, however, Picazo-Tadeo *et al.*, 2009b or the abovementioned paper by Gassner *et al.*, 2009, which also evaluate the performance of water operators at the level of particular production factors). It is our belief that these results might be of interest to managers and policymakers responsible for policies aimed at regulating the water industry.

Finally, it is worth mentioning some limitations of our approach that also constitute lines for future research. In the first place, it would be worth extending our methodology to incorporate non-controllable inputs and/or other environmental factors in order to attain more precise evaluations of performance. In this regard, it would be highly advisable to account for the quality dimension in urban water service provision; as mentioned above, if the analysis disregards quality it may overlook the fact that cutting the number of workers could result in lower service quality. Furthermore, it would also be worthwhile to integrate social aspects of water provision such as affordability into our analysis; e.g., it may be the case that if private operators reduce labour usage it could result in a less satisfactory achievement of social objectives such as affordability. A second interesting line for future research would be extending our approach to the analysis of both technical efficiency and price efficiency, e.g., it might be possible for a particular operator to score highly in terms of price efficiency but poorly for technical efficiency or vice-versa, and also to the study of how supposed efficiency gains from privatisation are distributed. Third, our manuscript suggests some factors and/or institutional restrictions that could explain the superiority of the private technology in managing labour; testing these hypotheses empirically would provide managers and regulators with sound information helping them to improve both management and water policies. And finally, replicating our analyses with larger samples of Spanish water utilities as well as for other developed countries would also be very welcome addition to this field of research.

Appendix

Using DEA techniques, the mathematical program required to calculate the directional metadistance function of expression (3), where both labour and operational costs are proportionally reduced, for a decision unit k' in the sample is:

$$
\begin{array}{ll}
M\overrightarrow{D}^{k'}_{radial} = \quad Maximize_{\delta^{k'}_{radial}, z^k} \; \delta^{k'}_{radial} & \\
\text{subject to:} & \\
y^{k'}_m \leq \sum_{k=1}^{70} z^k \, y^k_m & \text{m= water delivered, population served} \\
x^{k'}_f \geq \sum_{k=1}^{70} z^k \, x^k_f & \text{f= delivery network} \\
(1-\delta^{k'}_{radial}) x^{k'}_{v_i} \geq \sum_{k=1}^{70} z^k \, x^k_{v_i} & \text{i= labour, operational costs} \\
z^k \geq 0 & \text{k = 1, ... , 70} \\
\sum_{k=1}^{70} z^k = 1 &
\end{array}
\qquad (33)
$$

with z^k representing the weighting of each management unit k in the sample making up the efficient metafrontier to which unit k' is compared.

Likewise, the DEA-based program needed to compute the directional metadistance function of expression (4) for decision unit k' when only variable input i –either labour or operational costs– is reduced, while maintaining the other variable input –i and the vector of outputs is:

$$
\begin{array}{ll}
M\overrightarrow{D}^{k'}_{i} = \quad Maximize_{\delta^{k'}_{i}, z^k} \; \delta^{k'}_{i} & \\
\text{subject to:} & \\
y^{k'}_m \leq \sum_{k=1}^{70} z^k \, y^k_m & \text{m= water delivered, population served} \\
x^{k'}_f \geq \sum_{k=1}^{70} z^k \, x^k_f & \text{f= delivery network} \\
(1-\delta^{k'}_{i}) x^{k'}_{v_i} \geq \sum_{k=1}^{70} z^k \, x^k_{v_i} & \text{i } \epsilon \text{ n = labour, operational costs; and i} \notin -\text{i} \\
x^{k'}_{v_{-i}} \geq \sum_{k=1}^{70} z^k \, x^k_{v_{-i}} & -\text{i } \epsilon \text{ n = labour, operational costs} \\
z^k \geq 0 & \text{k = 1, ... , 70} \\
\sum_{k=1}^{70} z^k = 1 &
\end{array}
\qquad (34)
$$

In programs (33) and (34) variable returns to scale have been imposed by restricting the sum of the elements of the intensities vector, i.e., the weightings of decision units in the sample in the composition of the efficient frontier, to be equal to one (Banker *et al.*, 1984). The reason we do so is that we want to assess differences of efficiency specifically due to different managerial capabilities of public and private operators as well as differences between the two production technologies, but not differences due to the scale of operation.

Finally, formulating the programs needed to calculate the directional metadistance and distance functions of operators in group h requires only a few changes in notation and the substitution of the whole sample of decision units with units in group h, either public or private management, which is left to readers.

References

Aigner, D., Lovell, C.A.K., and Schmidt, P. (1977). Formulation and estimation of stochastic frontier production functions models. Journal of Econometrics 6: 21–37.

Ashton, J.K. (2000). Total factor productivity growth and technical change in the water and sewerage industry. The Service Industries Journal 20(4): 121–130.

Banker, R., Charnes, R., and Cooper, W. (1984). Some models for estimating technical and scale inefficiencies in Data Envelopment Analysis. Management Science 30: 1078–1092.

Barrera-Osorio, F., Olivera, M., and Ospino, C. (2009). Does society win or lose as a result of privatisation? The case of water sector privatisation in Colombia. Economica, 76(304): 649–674.

Bel, G., and Fageda, X. (2007). Why do local governments privatise public services? A survey of empirical studies. Local Government Studies 33(4): 517–534.

Bel, G., and Fageda, X. (2009). Factors explaining local privatization: a meta-regression analysis. Public Choice 139(1): 105–119.

Bel, G., Fageda, X., and Warner, M.E. (2010). Is private production of public services cheaper than public production? A meta-regression analysis of solid waste and water services. Journal of Policy Analysis and Management 29(3): 553–577.

Bel, G., and Warner, M.E. (2008). Does privatization of solid waste and water services reduce costs? A review of empirical studies. Resources Conservation and Recycling 52(12): 1337–1348.

Beltrán-Esteve, M. (2013). Assessing technical efficiency in traditional olive grove systems: A directional metadistance function approach. Economía Agraria y Recursos Naturales 13: 53–76.

Beltrán-Esteve, M., Gómez-Limón, J.A., Picazo-Tadeo, A.J., and Reig-Martínez, E. (2014). A metafrontier directional distance function approach to assessing eco-efficiency. Journal of Producitivity Analysis 41: 69–83.

Benito, B., Bastida, F., and García, J. (2010). Explaining differences in efficiency: An application to Spanish municipalities. Applied Economics 42(4): 515–528.

Berg, S.V., and Marques, R.C. (2011). Quantitative studies of water and sanitation utilities: a benchmarking literature survey. Water Policy 13(5): 591–606.

Bhattacharyya, A., Harris, T.R., Narayanan, R., and Raffie, K. (1995a). Specification and estimation of the effect of ownership on the economic efficiency of the water utilities. Regional Science and Urban Economics 25: 759–784.

Bhattacharyya, A., Harris, T.R., Narayanan, R., and Raffiee, K. (1995b). Technical efficiency of rural water utilities. Journal of Agricultural and Resource Economic 20(2): 373–391.

Bhattacharyya, A., Parker, P., and Raffiee, K. (1994). An examination of the effect of ownership on the relative efficiency of public and private water utilities. Land Economics 70(2): 197–209.

Botero, J.C., Djankov, S., La Porta, R., Lopez-de-Silanes, F., and Shleifer, A. (2004) The regulation of labor. Quarterly Journal of Economics 119(4): 1339–1382.

Bruggink, T.H. (1982). Public versus regulated private enterprise in the municipal water industry: A comparison of operating costs. Quarterly Review of Economics and Business 22: 111–125.

Byrnes, P., Grosskopf, S., and Hayes, K. (1986). Efficiency and ownership: Further evidence. Review of Economics and Statistics 68: 337–341.

Carvalho, P., Marques, R.C., and Berg, S. (2012). A meta-regression analysis of benchmarking studies on water utilities market structure. Utilities Policy, 21: 40–49.

Chambers, R., Chung, Y., and Färe, R. (1998). Profit, directional distance functions and Nerlovian efficiency. Journal of Optimization Theory and Applications 98: 351–364.

Charnes, A., Cooper, W.W., and Rhodes, E. (1978). Measuring the efficiency of decision making units. European Journal of Operational Research 2: 429–444.

Chong, E., Huet, F., Saussier, S., and Steiner, F. (2006). Public-private partnerships and prices: Evidence from water distribution in France. Review of Industrial Organization 29:149–169.

Coase, R. (1937) The nature of the firm. Economica 4(16): 386–405.

Conover, W.J. (1999). Practical nonparametric statistics. 3rd Edition. West Sussex: John Wiley & Sons.

Cook, W.D., and Seiford, L.M. (2009). Data envelopment analysis (DEA) — thirty years on. European Journal of Operational Research 192: 1–17.

Cooper, W.W., Seiford, L.M., and Tone, K. (2007). Data Envelopment Analysis. A comprehensive text with models, applications, references and DEA-Solver software. Springer, New York.

Crain, W.M., and Zardkoohi, A. (1978). A test of the property-rights theory of the firm: Water utilities in the United States. Journal of Law and Economics 21: 395–408.

Da Cruz, N.F., Marques, R.C., Romano, G., and Guerrini, A. (2012). Measuring the efficiency of water utilities a cross-national comparison between Portugal and Italy. Water Policy 14(5): 841–853.

Donahue, J. (1989). The privatization decision. Public Ends, Private Means, New York.

Emmenegger, P. (2011). Job security regulations in Western democracies: A fuzzy set analysis. European Journal of Political Research 50(3): 336–364.

Erbetta, F. and Cave, M. (2007). Regulation and efficiency incentives: Evidence from the England and Wales water and sewerage industry. Review of Network Economics 6(4): 425-52.

Estache, A., and Kouassi, E. (2002). Sector organization, governance and the inefficiency of African water utilities. World Bank Policy Research Working Paper 2890.

Estache, A., and Rossi, M.A. (2002). How different is the efficiency of public and private water companies in Asia?. The World Bank Economic Review 16(1): 139–148

Estache, A., and Trujillo, L. (2003). Efficiency effects of 'privatization' in Argentina's water and sanitation services. Water Policy 5(4): 369–380.

Färe, R., and Grosskopf, S. (2000). Theory and application of directional distance functions. Journal of Productivity Analysis 13: 93–103.

Färe, R., Grosskopf, S., and Lovell, C.A.K. (1994). Production frontiers. Cambridge, Cambridge University Press.

Faria, C.R., Souza, G., and Moreira, T. (2005). Public versus private water utilities. Empirical evidence for Brazilian companies. Economics Bulletin 8(2): 1–7.

Farrell, M. (1957). The measurement of productive efficiency. Journal of the Royal Statistical Society. Series A 120(3): 150–162.

Feigenbaum, S., and Teeples, R. (1983). Public versus private water delivery: A hedonic cost approach. Review of Economics and Statistics 65: 672–678.

Galiani, S., Gertler, P., and Schargrodsky, E. (2005). Water for life: The impact of the privatization of water services on child mortality. Journal of Political Economy, 113(1): 83–120.

García-Sánchez, I.M. (2006). Efficiency measurement in Spanish local government: The case of municipal water services. Review of Policy Research 23(2): 355–372.

García-Valiñas, M.A., González-Gómez, F., and Picazo-Tadeo, A. (2013). Is the price of water for residential use related to provider ownership? Empirical evidence from Spain. Utilities Policy 24: 59–69.

Gassner, K., Popov, A., and Pushak, N. (2009). Does private sector participation improve performance in electricity and water distribution? PPIAF Trends and Policy Options 6. Washington, DC: The World Bank.

González-Gómez, F., García-Rubio, M.A., Alcalá-Olid, F., and Ortega-Díaz, M.I. (2013). Outsourcing and efficiency in the management of rural water services. Water Resources Management 27(3): 731–747.

González-Gómez, F., Guardiola, J., and Ruiz-Villaverde, A. (2009). Reconsidering privatisation in the governance of water in Spain. Municipal Engineer 162: 159–164.

González-Gómez, F., Picazo-Tadeo, A.J., and Guardiola, J. (2011). Why do local governments privatise the provision of water services? Empirical evidence from Spain. Public Administration 89(2): 471-492.

Guerrini, A., Romano, G., and Campedelli, B. (2011). Factors affecting the performance of water utility companies. International Journal of Public Sector Management 24(6): 543–566.

Hall, D., Lobina, E., and de la Motte, R. (2005). Public resistance to privatisation in water and energy. Development in Practice 15(3-4): 286–301.

Hall, D., Lobina, E., and Terhorst, P. (2013). Re-municipalisation in the early 21st century: water in France and energy in Germany. International Review of Applied Economics 27(2): 193–214.

Hart, O., Shleifer, A., and Vishny, R. (1997). The proper scope of government: Theory and an application to prisons. Quarterly Journal of Economics 112(4): 1127–1161.

Hon, L.Y., Boon, T.H., and Lee, C. (2014) Productivity, efficiency and privatization in the Malaysian water industry. Journal of Southeast Asian Economies 31(2): 292–312.

Kirkpatrick, C., Parker, D., and Zhang, Y.F. (2006). An empirical analysis of state and private sector provision of water services in Africa. The World Bank Economic Review 20(1): 143–163.

Lambert, D.K., Dichev, D., and Raffiee, K. (1993). Ownership and sources of inefficiency in the provision of water services. Water Resources Research 29(6): 1573–1578.

Lannier, A., and Porcher, S. (2014). Efficiency in the public and private French water utilities: prospects for benchmarking. Applied Economics 46(5): 556–572.

Li, Q. (1996). Nonparametric testing of closeness between two unknown distribution functions. Econometric Reviews 15(3): 261–274.

Liu, J., Lu, L., Lu, W., and Lin, B.(2013). A survey of DEA applications. Omega 41, 893–902.

Lo Storto, C. (2013). Are public-private partnerships a source of greater efficiency in water supply? Results of a non-parametric performance analysis relating to the Italian industry. Water 5(4): 2058–2079.

Lobina E (2013) Remediable institutional alignment and water service reform: Beyond rational choice. Int Journal of Water Governance 1(1–2): 109–132.

Lobina, E., and Hall, D. (2000). Public sector alternatives to water supply and sewerage privatization. Case Studies. International Journal of Water Resources Development 16(1): 35–55.

Lobina, E., and Hall, D. (2001). U.K. water privatization - A briefing. Public Services International Research Unit, Greenwich.

Lobina, E., Kishimoto, S., and Petitjean, O. (2014). Here to stay: Water remunicipalisation as a global trend. Public Services International Research Unit (PSIRU), Transnational Institute (TNI) and Multinational Observatory, UK.

Mann, P.C., and Mikesell, J.L. (1976). Ownership and water system operation. Water Resources Bulletin 12(5): 995–1004.

Marin, P. (2009). Public-private partnerships for urban water utilities: A review of experiences in developing countries. World Bank Publications, Washington.

Maziotis, A., Saal, D., Thanassoulis, E., and Molinos-Senante, M. (2016). Price-cap regulation in the English and Welsh water industry: A proposal for measuring productivity performance. Utilities Policy, in press.

Mbuvi, D. (2012). Utility reforms and performance of the urban water sector in Africa. PhD Dissertation. Maastricht University.

Meeusen, W., and van den Broeck, J. (1977). Efficiency estimation from Cobb-Douglas production functions with composed error. International Economic Review 18(2): 435–44.

Meier, K.J., and O'Toole, L.J. (2011). Comparing public and private management: Theoretical expectations. Journal of Public Administration Research and Theory 21: 283–299.

Ménard, C., and Saussier, S. (2000). Contractual choice and performance the case of water supply in France. Revue d'Économie Industrielle, Programme National Persée, 92(1), 385–404.

Morgan, W.D. (1977). Investor owned vs. publicly owned water agencies: An evaluation of the property rights theory of the firm. Water Resources Bulletin 13(4): 775–781

Niskanen, W.A. (1971) Bureaucracy and representative government. Chicago: Aldine.

O'Donnell, C., Rao, D., and Battese, G. (2008). Metafrontier frameworks for the study of firm-level efficiencies and technology ratios. Empirical Economics 34(2): 231–255.

OECD (2003). Social issues in the provision and pricing of water services. OECD Publishing, Paris.

Peda, P., Grossi, G., and Liik, M. (2013). Do ownership and size affect the performance of water utilities. Evidence from Estonian municipalities. Journal of Management and Government 17(2): 237–259.

Pérard, E. (2009). Water supply: Public or private? An approach based on cost of funds, transaction costs, efficiency and political costs. Policy and Society 27: 193–219.

Picazo-Tadeo, A., Sáez-Fernández, F.J., and González-Gómez, F. (2008). Does service quality matter in measuring the performance of water utilities. Utilities Policy 16: 30–38.

Picazo-Tadeo, A.J., Castillo, J., and Beltrán-Esteve, M. (2014). An intertemporal approach to measuring environmental performance with directional distance functions: greenhouse gas emissions in the European Union. Ecological Economics 100: 173–182.

Picazo-Tadeo, A.J., González-Gómez, F., Guardiola, J., and Ruiz-Villaverde, A. (2012). Do ideological and political motives really matter in the public choice of local services management? Evidence from urban water services in Spain. Public Choice 151: 215–228.

Picazo-Tadeo, A.J., González-Gómez, F., and Sáez-Fernández, F.J. (2009a). Accounting for operating environments in measuring water utilities' managerial efficiency. The Service Industries Journal 29(6): 761–773.

Picazo-Tadeo, A.J., Sáez-Fernandez, F.J., and Gonzalez-Gomez, F. (2009b). The role of environmental factors in water utilities' technical efficiency. Applied Economics 41(5): 615–628.

Picazo-Tadeo, A.J., Sáez-Fernández, J., and González-Gómez, F. (2011). Assessing performance in the management of the urban water cycle. Water Policy 13: 782–796.

Pigeon, M., McDonald, D.A., Hoedeman, O., and Kishimoto, S. (2012). Remunicipalisation: Putting water back into public hands. Amsterdam, Transnational Institute.

Pigou, A. (1932), The economics of welfare. Macmillan & Co., London.

Romano, G., and Guerrini, A. (2011). Measuring and comparing the efficiency of water utility companies: A data envelopment analysis approach. Utilities Policy 19: 202–209.

Romano, G., Guerrini, A., and Vernizzi, S. (2013). Ownership, investment policies and funding choices of Italian water utilities: An empirical analysis. Water Resources Management 27(9): 3409–3419.

Ruíz-Villaverde, A., González-Gómez, F., and Picazo-Tadeo, A.J. (2015). The privatisation of urban water services: Theory and empirical evidence in the case of Spain. Investigaciones Regionales 31: 157–174.

Saal, D.S., and Parker, D. (2004). The comparative impact of privatization and regulation on productivity growth in the English and Welsh water and sewerage industry: 1985-1999. International Journal of Regulation and Governance 4(2): 139–70.

Saal, D.S., and Parker, D. (2000). The impact of privatization and regulation on the water and sewerage industry in England and Wales: A translog cost function model. Managerial and Decision Economics 21: 253–268.

Saal, D.S., and Parker, D. (2001). Productivity and price performance in the privatized water and sewerage companies of England and Wales. Journal of Regulatory Economics 20(1): 61–90.

Saal, D.S., Parker, P., and Weyman-Jones, T.G. (2007). Determining the contribution of technical change, efficiency change and scale change to productivity growth in the privatized English and Welsh water and sewerage industry: 1985–2000. Journal of Productivity Analysis 28: 127–139.

Sabbioni, G. (2008). Efficiency in the Brazilian sanitation sector. Utilities Policy 16: 11–20.

Sáez-Fernández, F., Picazo-Tadeo, A.J., and Llorca-Rodríguez, C. (2012). Do labour societies perform differently to cooperatives? Evidence from the Spanish building industry. Annals of Public and Cooperative Economics 83(1): 61–81.

Shephard, W. (1970). Theory of cost and production functions. Princeton University Press, Princeton.

Simar, L., and Zelenyuk, V. (2006). On testing equality of distributions of technical efficiency scores. Econometric Reviews 25(4): 497–522.

Thomas, S., Hall, D., Lethbridge, J., Lobina, E., Popov, V., Corral, V., and van Niekerk, S. (2012) Why water is a public service: Exposing the myths of privatization. Public Services International Research Unit, University of Greenwich, London.

Williamson, O. (1976). The economics of internal organization: Exit and voice in relation to markets and hierarchies. American Economic Review 66(2): 369–77.

Zschille, M., and Walter, M. (2012). The performance of German water utilities: A (semi)-parametric analysis. Applied Economics 44(29): 3749–3764.

Conclusions

In a context of increasing scarcity of water resources, the main objective of the present doctoral dissertation has been to add to the knowledge base on several aspects related to urban water management. In broad terms, the main conclusions derived from the essays included can be grouped around four issues: the implications for demand-side policies; recommendations for governance in urban water management; methodological contributions; and policy implications for water management in Spain.

Demand-side policies

As outlined in the introduction, demand-side policies, both pricing and non-pricing, play an ever more essential role in addressing problems related to water scarcity, urban water sustainability and service continuity. Against this background, Essays 1 and 2 of this dissertation are aimed at contributing to the design and implementation of appropriate tariffs and pricing policies. In this respect, in Essay 1 we find that, contrary to widely-held assumptions in the residential water demand literature, water consumption is neither linear in income nor separable from the rest of the goods consumed in the household and taking those aspects into account may lead to more accurate estimates of demand patterns. This has important implications, as those estimates are the foundation for designing and predicting the effects of pricing policies.

Likewise, our results in Essay 2 suggest that price-setting decisions in urban water services may often be influenced not only by economic issues and factors involving the environment in which the service is supplied, but also by political and ideological factors, as well as rent-seeking strategies by decision-makers and politicians. As this may affect the capacity of policymakers to set prices that can ensure the simultaneous achievement of multiple water management objectives, such as efficiency, sustainability and affordability, mechanisms should be established in order to limit its interference in decision-making. Accordingly, in countries where they are not already in place, the establishment of supervisory bodies, as well as an adequate regulation, control and monitoring, emerge as key factors for the improvement of pricing policies.

Finally, results from Essay 3 seemed to indicate that certain pricing and non-pricing policies related to residential water management that are becoming more widely implemented around the world may affect other behaviours adopted by the household, with unintended environmental and economic consequences. Thus, a thorough cost-benefit analysis of the corresponding policies should incorporate those consequences. In addition, promoting pro-environmental habits was found to be an efficient means of preventing mitigating behaviours that generate negative environmental externalities. Furthermore, we found that an improvement in tap water quality may be an effective tool to reduce the use of bottled water.

Efficiency and governance in urban water management.

Residential water management also requires improved governance of urban water services, which can be achieved by designing more efficient management systems. In this sense, the type of ownership of the management of the service is usually acknowledged as playing an important role in explaining the efficiency of the operator. However, the literature has not reached a consensus as to which form of ownership performs better. In this context, our results seemed to suggest that each ownership type is best at managing one or several of the production factors. Likewise, the factor that is best managed may in turn depend on the particular institutional framework and technological restrictions faced by the different forms of ownership. In that case, synergies may be achieved by combining management systems and implementing the institutional and regulatory backgrounds behind the most efficient management of each production factor (Essay 4).

Methodological contributions

Policy making should be supported by the development of metrics and methodological approaches to inform the implemented policies. Therefore, this dissertation is also aimed at meeting that need. Particularly, in Essay 1 we implement a system of demand estimation that is novel in water demand analysis.

Furthermore, we found that it displays a better fit to the observed demand patterns for residential water consumption than the ones previously employed. Therefore, it seems to offer a more precise tool to determine price and income elasticities, and helps improve predictions of the welfare effects of pricing policies and taxes. Moreover, this new development can potentially be applied in a substantial number of countries that currently rely on country-representative Household Budget Surveys, and can thus inform national policies related to residential water.

Essay 2 filled a gap in the literature by designing a metric that can be used to measure the level of escalation embedded in water tariffs and that permits full comparability among municipalities regardless of the specific tariff structure they use; i.e., regardless of the number and size of price blocks, the presence and size of fixed components, and price levels included in the tariffs. This measure can be of help when it comes to monitoring and controlling the impact of policies affecting the structure of the tariff.

In Essay 3, a methodological strategy was proposed to deal with the substantial number of zero consumption records in databases on averting behaviours. We showed that failure to address this problem properly may lead to bias in the estimates of the effects of the policies aimed at controlling these behaviours.

Finally, Essay 4, also added a methodological contribution, by proposing a novel methodological approach combining Data Envelopment Analysis (DEA), metafrontiers and directional distance functions, which proved successful in providing new insights about the relationship between ownership of the service provider and efficiency in the urban water sector. We believe that this methodological approach can assist policy makers in making better decisions with respect to the choice of ownership type. Furthermore, it can help identify and implement the regulatory frameworks and technologies that prove to be most effective at efficiently managing the different production factors. Moreover, it could be extended to similar analyses that seek to compare the performance of two production technologies in relation to the efficiency of the service.

Policy lessons for water management in Spain

Finally, given that the empirical setting of this dissertation is Spain, some conclusions can be drawn that are specific to water management in this country. Particularly, we found that there is still plenty of room for improvement when it comes to pricing policies. Following the stipulations set out in the Water Framework Directive (WFD), tariffs that reflect the full cost of the service (full-cost recovery) as well as environmental and other costs related to resource use, should be implemented. Moreover, pricing policies are expected to promote an efficient use of the resource. In order to simultaneously achieve the abovementioned objectives, the general recommendation is for tariffs in increasing block rates, which embed a certain level of escalation in prices. However, our results in Essay 2 seemed to indicate that following the implementation of the WFD, there has been a decrease in the degree of price escalation in the variable component of the tariff in Spain, contrary to what was expected. We also found that when we include the fixed component of the tariff in the analysis, tariffs in Spain are in fact regressive on consumption, albeit less so in the years after the entry into force of the WFD. Therefore, a more ambitious implementation of the WFD should be promoted, designing policies that ensure the continuity of the service through adequate funding and maintenance of the distribution networks, and tariffs that promote universal access, efficiency and sustainability of water resources.

Moreover, in this dissertation we highlighted that the lack of a regulatory body that establishes common guidelines lead to failings in price-setting strategies, thus hindering the application of satisfactory water management solutions in a country suffering from water stress throughout most of its territory. More specifically, some concern is raised about the desirability of making decisions at the municipal level that can affect a wider region such as a river basin. Independently setting tariffs that promote a sustainable use of the resource cannot be effective if other agents that share the same source of abstraction do not align their policies in the same direction. Thus, some standards and controls should be established, at least at the watershed level. Moreover, we found that besides more pragmatic reasons related to the environment in which the service is supplied, other factors such as the ideology of the government or the ownership of the management also affect pricing decisions. We believe that a regulatory body such as the one described above would be extremely useful in preventing political interference and arbitrariness in price-setting strategies.

Conclusiones

En el contexto actual de creciente escasez de los recursos hídricos, el principal objetivo de esta tesis doctoral es contribuir a la mejora de las políticas urbanas de gestión del agua. En un sentido amplio, las conclusiones centrales derivadas de los ensayos incluidos en la misma se pueden agrupar en torno a cuatro aspectos. En concreto, implicaciones para las políticas de demanda y para la gobernanza en la gestión urbana del agua, contribuciones metodológicas, y lecciones de política pública para la gestión del agua en España.

Políticas de demanda.

Tal como se ha abordado de forma más detallada en la introducción, las políticas de demanda, tanto tarifarias como no tarifarias, juegan un papel cada vez más esencial a la hora de abordar los problemas relacionados con la escasez de recursos hídricos, la sostenibilidad del servicio urbano de agua y la continuidad en su provisión. En este contexto, el primer y segundo ensayo de esta disertación tienen por objeto realizar aportaciones al adecuado diseño e implementación de tarifas y políticas de demanda. A este respecto, los resultados del primer ensayo parecen sugerir que, contrariamente a lo que se ha venido asumiendo de forma habitual en la literatura de demanda residencial de agua, el consumo de agua no es ni lineal en el ingreso ni separable del resto de bienes consumidos en el hogar, y que la consideración de estos aspectos permitiría obtener estimaciones más precisas de los patrones de demanda observados. Esto tiene importantes implicaciones en términos de política pública, ya que estas estimaciones son la base para el diseño y predicción de los efectos de las políticas tarifarias.

En una línea similar, en el segundo ensayo se encuentra que en muchas ocasiones las decisiones de fijación de precios en los servicios urbanos de agua pueden estar influenciadas no únicamente por aspectos económicos o factores relacionados con las características ambientales en las que se presta el servicio, sino también por cuestiones políticas, ideológicas y otros factores relacionados con estrategias de captura de rentas por parte de políticos y responsables de la formulación de políticas públicas. Esto podría estar influenciando la capacidad de los responsables de la formulación de dichas políticas para fijar precios que sean capaces de asegurar que se alcancen de forma simultánea los múltiples objetivos a los que se espera que responda la gestión de los recursos hídricos, como eficiencia, sostenibilidad y asequibilidad. Por tanto, se deberían de establecer mecanismos que limiten su interferencia en la toma de este tipo de decisiones. Bajo estas circunstancias, la existencia de un órgano supervisor, así como una debida regulación, control y monitoreo, se revelan factores fundamentales para la mejora de las políticas tarifarias en aquellos países en los que aún no se hallen establecidos.

Por último, los resultados del tercer ensayo parecen indicar que algunas políticas tarifarias y no tarifarias relacionadas con la gestión de los recursos hídricos a nivel residencial que se están aplicando cada vez con mayor frecuencia en muchas partes del mundo podrían estar afectando a otros comportamientos llevados a cabo en el hogar (e.g. consumo de agua embotellada), con consecuencias inesperadas en términos económicos y medioambientales. Por tanto, un análisis coste-beneficio exhaustivo de las correspondientes políticas debería incorporar dichas consecuencias. Además, se encuentra que promover hábitos pro-medioambientales podría prevenir de forma eficaz la realización de este tipo de comportamientos defensivos que implican externalidades medioambientales negativas. Asimismo, una mejora en la calidad del agua del grifo parece mostrarse efectiva como forma para reducir el uso del agua embotellada.

Eficiencia y gobernanza en la gestión del agua urbana.

La gestión del agua residencial exige asimismo mejoras en la gobernanza de los servicios urbanos de agua a través del diseño de sistemas más eficientes que se demuestren más resilientes a la cada vez más reducida disponibilidad del recurso. En este sentido, es habitualmente reconocido en la literatura que la titularidad del gestor del servicio juega un papel importante en cuanto a lo que la eficiencia del operador se refiere. Sin embargo, no ha sido posible alcanzar un consenso respecto a qué forma de propiedad se muestra superior en términos de desempeño. Bajo este contexto, nuestros resultados parecen sugerir que podría ser que cada tipo de propiedad del gestor sea capaz de gestionar mejor uno o varios de los factores productivos. De igual modo, qué factor resulta mejor gestionado podría estar dependiendo del marco institucional y las restricciones tecnológicas afrontadas por cada forma de titularidad. En este caso, se podrían conseguir sinergias propiciando la combinación de varios sistemas de gestión e implementando aquellos marcos institucionales y regulatorios que se muestren más exitosos a la hora de gestionar cada factor de producción de forma eficiente (*Ensayo 4*).

Contribuciones metodológicas.

La formulación de políticas públicas debe apoyarse en el desarrollo de métricas y enfoques metodológicos que puedan informarlas adecuadamente. Por tanto, esta disertación trata de contribuir también a esta área. En concreto, en el primer ensayo se implementa un sistema de demanda que no ha sido utilizado con anterioridad en el análisis de la demanda de agua. Encontramos que este sistema ajusta mejor a los patrones observados de demanda para demanda de agua residencial que aquellos empleados de forma mayoritaria con anterioridad. De este modo, parece ofrecer una herramienta más precisa a la hora de determinar las elasticidades precio e ingreso y para la mejora de las predicciones de los efectos de las políticas tarifarias y de impuestos sobre el bienestar social. Asimismo, esta mejora podría ser aplicada de forma potencial a un elevado número de países que cuentan actualmente con Encuestas de Presupuestos Familiares representativas de la población como las utilizadas en este trabajo, con el objeto de informar de forma más precisa la toma de decisiones y políticas públicas relacionadas con la gestión de los recursos hídricos a nivel nacional.

De igual modo, el segundo ensayo cubre un vacío existente en la literatura diseñando una métrica que permite medir el nivel de escalado de las tarifas de agua y comparar entre jurisdicciones independientemente de tipo de tarifa empleado –es decir, del número y tamaño de los bloques, presencia y tamaño del componente fijo, y niveles de precios generales del municipio. Esperamos que esta medida sea capaz de dar soporte a las labores de control y monitoreo de las políticas que afecten al diseño de estructuras tarifarias. En el tercer ensayo, se propone una estrategia metodológica que permite lidiar con la habitual aparición de una elevada proporción de respuestas nula (o cero) en la literatura de comportamientos defensivos. Mostramos que no abordar de forma adecuada este problema puede causar sesgo en las estimaciones de los efectos de las políticas que tienen por objeto controlarlos. Por último, el cuarto ensayo añade asimismo una contribución metodológica a través de una estrategia empírica novedosa en la medida de la eficiencia en el sector del agua que combina Análisis Envolvente de Datos (DEA), metafronteras y funciones de distancia direccionales, y que se muestra satisfactoria en ofrecer nuevas perspectivas en lo referente a la relación entre propiedad del gestor y eficiencia en el servicio urbano del agua. Consideramos asimismo que este enfoque metodológico podría servir de soporte a los responsables de política pública para tomar mejores decisiones respecto a la elección de la titularidad del gestor, así como a detectar e implementar aquellos marcos regulatorios y tecnologías que se demuestren más efectivas en la gestión eficiente de los diferentes factores de producción. Además, esta metodología podría ser extendida a análisis similares en los que se pretenda comparar la superioridad de dos tecnologías de producción en relación con la eficiencia en la prestación del servicio.

Lecciones de política pública para la gestión de los recursos hídricos en España.

Por último, dado que el análisis empírico de esta disertación se basa en el caso español, algunas de las conclusiones derivadas de la misma son de aplicación específica para la gestión del agua en este país. Particularmente, se encuentra que las políticas de precios en España cuentan aún con un elevado margen de mejora. Siguiendo las indicaciones de la Directiva Marco del Agua (DMA), las tarifas deberían reflejar el coste total del servicio (recuperación de costes), así como los costes medioambientales y de uso del recurso. De igual modo, sería de esperar que las tarifas promovieran un uso eficiente del mismo. Para alcanzar de manera simultánea estos objetivos, habitualmente se recomienda la utilización de tarifas en bloques crecientes de consumo, que implícitamente introducen un cierto nivel de escalado en el precio. Sin embargo, los resultados obtenidos en el segundo ensayo parecen indicar que, tras la implementación de la DMA en España, contrariamente a lo que sería de esperar se ha producido una caída en el grado de escalado en el precio dentro de la parte variable de la tarifa. Se encuentra así mismo que cuando se considera también el componente fijo, las tarifas en España son regresivas en el consumo, aunque se observa que el nivel de regresividad se ha reducido en los años que han seguido a la implementación de la DMA. Por tanto, se debería promover una trasposición más ambiciosa de la DMA que permita el diseño de políticas que aseguren la continuidad del servicio mediante una financiación adecuada y mantenimiento de las redes de distribución, así como tarifas que promuevan acceso universal, eficiencia y sostenibilidad de los recursos hídricos.

De igual modo, esta disertación pone en relieve que la ausencia de un organismo regulador que establezca unas directrices comunes puede estar causando disfunciones en la toma de decisiones relativas a la fijación de precios en el sector del agua. Esto, a su vez, podría estar dificultando la aplicación de soluciones que resulten satisfactorias para gestionar adecuadamente el recurso, en un país que se halla sometido a estrés hídrico en la mayor parte de su territorio. En concreto, cabría cuestionar si es razonable que decisiones que afectan en última instancia a un área mucho más amplia como es la cuenca hidrográfica sean tomadas a nivel municipal. Fijar de forma aislada tarifas que promuevan un uso sostenible del recurso puede ser de poca utilidad si el resto de agentes que comparten la misma fuente de captación del recurso no aplican políticas en la misma dirección. Por tanto, sería aconsejable establecer ciertos estándares y controles, al menos a nivel de la cuenca hidrográfica. Por otra parte, se encuentra que al margen de razones de corte pragmático relacionadas con el entorno en el que se provee el servicio, otros factores como la ideología del partido en el gobierno o la titularidad del gestor también pueden afectar a las decisiones tarifarias. Un organismo regulatorio como el descrito anteriormente podría mostrarse muy efectivo a la hora de evitar la existencia de injerencia política y arbitrariedad en las estrategias de fijación de precios en los servicios de agua urbana.