INTEGRATED WATER SUPPLY-DEMAND MODELLING INCLUDING DYNAMIC PRICING

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Executive Summary

This document is the Deliverable **D5.3, Integrated water supply-demand modelling including dynamic pricing**, which, according to the Annex I to the contract, has the following goals.

D5.3) Integrated water supply-demand modelling including dynamic pricing: This deliverable contains a review of water scarcity indicators: existing water scarcity indicators will be reviewed with focus on urban and peri-urban contexts and considering their potential to trigger pricing schemes. Then it reports on the water scarcity indicators developed for the two case studies and the analysis of how they can trigger different pricing schemes. Finally, it delivers a comparative analysis and recommendations: the water demand estimated by running the econometric water demand models will be comparatively analysed w.r.t. the baseline identified in T5.1.

This deliverable uses two case-studies, in London in the United Kingdom, and in Valencia in Spain, to explore the potential for smart metering to foster dynamic scarcity pricing of water, that is, a water price that changes according to scarcity conditions.

The main contributions of the deliverable on the design of dynamic scarcity pricing are as follows:

- The marginal resource opportunity cost (MROC), which is defined as the benefit forgone by diverting an additional unit of the resource away from its more productive use, is a universal indicator of water scarcity that also can serve as a basis for pricing instruments.
- Pricing water at its MROC when it is scarce is promoting efficient use of water throughout the river basin, one of the objectives of the Water Framework Directive.
- Urban scarcity pricing consists in adding the MROC of water to the volumetric charge that residential users usually have to pay.

The main findings of both case-studies are as follows:

- Applications to both London and Valencia, two case-studies with very different characteristics show the potential of dynamic pricing to reduce consumption when water becomes scarce.
- In both case-studies, price increases are almost exclusively in the 0-100% range. In Valencia, higher price increases are non-existent. In London, they are triggered by situations of extreme scarcity when dynamic pricing alone is no longer sufficient to deal with the crisis, and should be enforced alongside regulatory use restrictions. Such regulations would then replace the price increases.

Other main contributions from the London case-study are:

- The London case-study describes mid-21st century water scarcity caused by a growing population. It shows that scarcity pricing can help preserve environmental flows across a range of scenarios and cases defined by different set of economic parameters.
- The analysis shows the importance of evaluating those parameters to understand the potential benefits of the implementation of scarcity pricing. In particular, the demand response to price changes should be understood across a range of prices.
- The London case-study points out the limitation of scarcity pricing as a tool for preventing the depletion of rivers, especially in the case of extreme droughts. Then, economic principles must be complemented by strict policies regulating water use.

Other main contributions from the Valencia case-study are:

- The Valencia case-study outlines specific water tariffs indexed on reservoir levels in a river basin where the large storage capacity of reservoirs is the best buffer against drought; therefore, water in the reservoirs must be conserved when levels are lower than usual.
- Scarcity pricing is a relevant tool for utility to not run a deficit when water must be conserved.
- The Valencia case-study demonstrates the potential for scarcity pricing schemes to be applied as soon as water utilities feel this kind of tariff is no longer a taboo.

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1. Introduction

This deliverable analyses the relationship between water scarcity, scarcity indicators and scarcity-based dynamic pricing, using case-studies from the SmartH2O project. It focusses on the two case-studies where there is sufficient water stress to make this analysis relevant: the London area in the Thames River basin in the UK and the Valencia area in the Jucar river basin in Spain. It also focusses on the only form of dynamic pricing defined in D5.2 and MS10 that affects the balance between supply and demand: scarcity pricing. The impacts of time-of-day pricing, the other form of dynamic pricing, essentially concern a utility's finances, and have been covered by D8.5.

This deliverable is organised as follows.

Section 2 reviews existing indicators of urban water scarcity. It shows that urban water scarcity cannot be conceived separately from water scarcity in other economic sectors in the same basin. Then, water scarcity is quantified by a financial indicator, the marginal resource opportunity cost (MROC) of urban use. This is the maximal value that the last unit of water given to residential consumption would have had, if it had been given to other uses. According to microeconomic theory, there is a perfect market for a commodity when the MROC equals the price users are willing to pay for all uses. Therefore, pricing water at its MROC when it is scarce is promoting efficient use of water throughout the river basin, one of the objectives of the Water Framework Directive.

This deliverable uses hydro-economic modelling to represent and value water flows in the basin under the whole range of plausible hydro-climatic circumstances, including drought. Ideally, the MROC is computed through basin-wide optimisation, but this, besides being difficulty to carry out in many cases, does not mirror the actual use of water by all actors, whether urban or non-urban. Therefore, this deliverable uses simulation to compute the MROC in the London and Valencia cases, and propose dynamic tariffs adapted to the situation of the two cities.

Section 3 uses the notion of MROC to propose the theoretical foundation of dynamic scarcity pricing of urban water. For this, other economic concepts are introduced to describe the aggregate response of urban water users to price changes: the price elasticity of demand and the graphical representation of the relationship between price and demand through the demand curve. This visual tool is used to show how the volumetric price of water for urban consumption can be deduced from the MROC in order to promote water conservation and efficient management in scarcity situations.

This toolbox of economic concepts and pricing strategies has then been applied to the case studies, which are complementary because they present very different characteristics, which allows for exploring different aspects of scarcity pricing. Taken together, they show the potential for the implementation of dynamic scarcity pricing in a range of circumstances

Section 4 presents the London case-study and its results. London is part of the Thames River basin, where urban use dominates, and competes only with the imperative of keeping freshwater running in the river to ensure the provision of ecosystem services – including biodiversity protection, recreational uses and property valuation. Rising demand from the expanding city drives current and future water stress, and the potential for scarcity pricing is investigated using estimated mid-21st-century demands. Available storage represents only three months worth of urban consumption, so that this is a situation where droughts can happen quickly, and generally do not last very long. To mirror this, scarcity pricing has to adapt to quickly changing conditions, e.g., on a weekly timescale.

Since the case-study focusses on future water scarcity driven by a rising demand in an area where the volumetric pricing of water is a new idea for many users, the goal of the case-study is to understand the impact of scarcity pricing on the preservation of environmental flows under a range of economic parameters describing the value of water to both urban users and environmental flows. This is also motivated by the fact that those parameters, although necessary for the implementation of scarcity pricing, are largely unknown.

Section 5 presents the Valencia case-study and its results. Contrary to the situation in the Thames River basin, urban use only represents around 14% of consumption in the Jucar river basin where Valencia is situated, and where agriculture accounts for the bulk of the demand. Storage capacity is worth a year and a half of total water consumption in the basin, so that scarcity situations span over several months and can be forecast well in advance. Contrary to the Thames basin, where water scarcity is a somewhat new and growing concern, water management in the Jucar basin is the mirror of centuries of episodic drought. Due to these storage and institutional constraints, dynamic prices must be the result of well-established rules, and be fixed possibly for several months depending on reservoir levels.

Therefore, the Valencia case-study presents a rigorous framework where hydro-economic modelling is used to simulate the MROC of water in the system's reservoirs, and where the results then serve as a basis to design tariffs that are indexed on reservoir levels. This methodology could readily be applied in real-world situations, when and where water utilities and regulators feel ready to implement dynamic scarcity pricing to promote water conservation and usage efficiency. Indeed, it considers various aspects of water tariff design, such as equity, revenue sufficiency, or consideration of different socio-economic groups within the customer base. All of these factors are usually considered by decision-makers.

Finally, a conclusion section is presented to put the findings under perspective. A comparison of the two studies is presented to clarify the range of situations in which dynamic scarcity pricing could be useful, and the main results are summarised according to that comparison. This section also discusses some of the assumptions made in these assessments. Those include most notably assumptions on the price elasticity of water, but also on other economic and hydrological parameters of our simulations. How these data could be obtained in future assessments of dynamic scarcity pricing is also discussed.

2. Indicators of urban water scarcity

This section presents a review of indicators of water scarcity in an urban context. It starts with an overview of some common non-price indicators in Section 2.1. Then, it focuses on the evaluation of water scarcity by evaluating its marginal resource opportunity cost (MROC), that is, the economic benefit foregone by not committing the last unit of water to other uses (Section 2.2). The MROC of water is a universal economic measure of scarcity. This evaluation of scarcity through water price is performed through hydro-economic modelling, which will be a basis for introducing dynamic pricing in the rest of this deliverable.

2.1 Examples of non-price indicators

Urban water scarcity has been measured using diverse indicators. The aim of this section is to provide some instances of a subsample of them referred to as non-price indicators, that is, indicators that do not refer to the price of a commodity when it is scarce.

The City Blueprint, one of the actions of the European Innovation Partnerships (EIPs) on Water, provides a comprehensive overview of the state of the Integrated Water Resources Management (IWRM) in urban areas. The proposed format includes 24 indicators, which are meant to comprehensively describe the sustainability of urban water cycle services [Leeuwen et al., 2015]. These indicators are classified in 8 areas:

- Water security
- Water quality
- Drinking water
- Sanitation
- Infrastructure
- Climate robustness
- Biodiversity and attractiveness
- Governance

Among them, the area that is of particular interest in our review is water security, which in turn is based on three indicators:

- Total water footprint
- Water scarcity
- Water self-sufficiency

2.1.1 Water security

The total water footprint is the total volume of freshwater that is used to produce the goods and services consumed by the community. The water self-sufficiency, instead, is the ratio of the internal to the total water footprint (see [Hoekstra and Chapagain, 2007] and [Hoekstra et al., 2009]).

The water scarcity indicator comprises 3 sub-indicators:

- Freshwater scarcity
- Groundwater scarcity
- Salinization and seawater intrusion

The freshwater scarcity is the abstracted freshwater as percentage of total renewable resources, which includes surface water and groundwater sources. This percentage is in turn translated in a score in accordance with the European Environmental Agency classification.

Scoring method (EEA classification)	
% of renewable resources abstracted	**Score**

0-2	0
2-10	1
10-20	2
20-40	3
>40	4

Table 1. Scoring method – Freshwater scarcity

The data needed to compute the indicator are available via Aquastat (accessible on the website of the Food and Agriculture Organization of the United Nations - FAO).

Groundwater scarcity is defined as the ratio between the abstracted groundwater and the annual groundwater recharge. This is a measure of the pressure on groundwater resources. This percentage is in turn translated in a score in accordance with the UNESCO classification.

Scoring method (UNESCO classification)	
% abstracted of annual recharge	**Score**
0-2	0
2-20	1
20-50	2
50-100	3
>100	4

Table 2. Scoring method – Groundwater scarcity.

The data needed to compute the indicator are available via the Groundwater Development Stress (accessible on the website of the UNESCO).

The salinization and seawater intrusion indicator measures the vulnerability of the soil to seawater intrusion. The indicator is based on a literature check after which seawater and groundwater intrusions are scored as below. The highest score of both indicators is used as the final score for salinization and seawater intrusion.

Seawater intrusion	
Description	**Score**
No seawater intrusion reported and city not prone to (future) intrusion	0
No seawater intrusion reported and city can experience intrusion in coming century	1
No seawater intrusion reported but city is prone to intrusion in the near future	2
Seawater intrusion reported	3
Seawater intrusion reported and city is particularly prone to intrusion	4
Groundwater salinization	
Description	**Score**
No concern	0

Low concern	1
Medium concern	2
Concern	3
Great concern	4

Table 3. Scoring method – Salinization and seawater intrusion.

The data needed to compute the indicator are available via the European Environmental Agency (Indicator fact sheet) and via the Joint Research Centre: European soil portal – Soil data and information system.

2.1.2 Urban water scarcity and scarcity at river basin level

The use of global water scarcity indicators to measure the water scarcity at the city level is a common staple of the policy literature. It enables to draw comparison between cities and stresses the areas in which a city should act to abide by global standards. However, the impact of water scarcity on cities (that is, whether citizens experience water scarcity) is correlated to the rate of change in water availability in the river basin they draw water from [Rijsberman and Mohamed, 2003]. This means that, even as the city itself is the focal point of attention from decision-makers, the relevant decisions to alleviate a city's water scarcity and water security problems are taken at the basin level.

From an adaptation perspective, a relevant variable is the path through which water availability decreases in a given area [Rijsberman, 2006]. It is well recognized that where water scarcity at the river basin level is constantly high or changes slowly, systems of water use (at the city level) can adapt to those conditions. Water institutions act more readily in response to the local water scarcity context when conditions evolve more slowly.

In contrast, rapidly increasing water scarcity requires water users and water institutions to adapt to new scarcity conditions. The adaptation process to the new scarcity context can be long and problematic. With increasing competition and lagging institutions, conflicts can arise. Typical conflicts are those between rapidly growing urban areas or conflicts between agriculture and the environment.

Regardless of the pace at which conditions evolve, putting a price tag on the water supply sources city draw water from is best achieved through economic indicators of water scarcity. The next paragraph will detail a type of indicator that runs through the water resources literature.

2.2 The marginal resource opportunity cost (MROC)

2.2.1 Definition

The marginal resource opportunity cost (MROC) is the benefit forgone by not allocating an additional unit of water introduced to its most productive use. It has been described as expressing "the basic relationship between scarcity and choice" [Buchanan, 2008]. Indeed, if there is no scarcity, all demands are satisfied in the sense that there is enough of the resource to allocate it to all its productive uses. When scarcity arises, not all demands can be satisfied simultaneously, and a limited amount of resource must be allocated. Then, the MROC measures the economic performance of a given allocation mechanism, i.e., of the way choices are made in order to allocate limited resources.

A distinction must be made between gross MROC and net MROC. On one hand, the gross MROC corresponds to the net value of allocating an additional unit of resource to its most productive use. It does not depend on what the alternative uses are. Resource allocation is efficient when the gross MROC is the same for all uses, i.e., allocating an additional unit of resource to all uses leads to the same result and there is no economic incentive to choose any one use over the others. On the other hand, the net MROC represents the difference between the value of allocating a unit of resource to a given use and its value if it were allocated to its more productive use. It is the benefit forgone by allocating a unit of resource away from its more productive and towards an another use. It therefore depends on this other use. Resource allocation is efficient when the net MROC is zero for all uses.

2.2.2 Specificities of the MROC of water

The MROC is an economic indicator that can be called universal, in the sense that it can be applied to any resource in any situation, and that it accounts for all the uses of a resource from the point of view of the economic value of these uses. From this point on, this deliverable will exclusively focus on the gross MROC of water, which will simply be called MROC of water. The MROC of water is defined at a precise point in space and time [Harou et al., 2009]. Spatial location matters, because water flow is directional, so a unit of water available at any given location cannot be used upstream of that location. Besides, water scarcity varies strongly with time, as water demand in a river basin often follows a seasonal pattern, while water availability does as well. In many instances, demand tends to be highest in summer, at a time when water availability is low. Another important specificity of the MROC of water is that it evolves on weekly or longer timescales, which is due to the fact that water systems typically have large storing capacity.

In that respect, one needs to distinguish between the value of water for immediate use, and its value for future use. In reservoirs that can store water, the MROC takes into account the benefits that can be reaped from future water use. Efficient allocation must allocate water through time as well as allocating it at present between actors. Thus, the concept of hedging in water resources [Draper and Lund, 2004] introduces the idea that in some situations it is better to artificially increase water scarcity in the present by keeping it for future use, than to risk severe water shortage in the future. [You and Cai, 2008] carry this analysis further by showing that present MROC must equate the expected value of the future MROC for hedging to be optimal, and by showing that this equation is made more complicated by physical factors – such as evaporation from reservoirs – and by economic factors – such as risk aversion.

Therefore, in order to account for future uncertain water supplies, reservoirs are where the MROC of water is measured [Tilmant et al., 2008; Pulido-Velazquez et al., 2013], and this includes surface water reservoirs but also groundwater reservoirs [Pulido-Velazquez et al., 2008]. The MROC of water is also spatially variable, which brings a word of caution about treating an aquifer as a spatially homogeneous reservoir in the same way as a surface water reservoir, especially since water also flows directionally in an aquifer [Brozovic et al., 2010]. For both types of reservoir, the MROC is linked with reservoir levels and current and future water availability and demand.

In summary, here are the different ways to define the MROC of water, which reflects water scarcity:

- For water allocation at a given period of time, the MROC is the benefit forgone by not allocating water to its most productive current use.
- The MROC of water in a reservoir reflects its (expected) future value, that is, the (expected) value of an additional unit of water affected to the most productive future use.

Note that these two definitions are in fact related, as the most productive current use of water can be to store it in a reservoir for future use. Besides, recall that these economic indicators of MROC reflect water scarcity for all uses simultaneously, and that includes urban use.

2.2.3 Evaluation through hydro-economic modelling

The MROC of water usually cannot be accessed through water market, because fully functioning water markets are very rare [Timant et al., 2008]. In fact, because of the directional nature of water flows, market design is fraught with difficulties if the aim is to establish an economically efficient marketplace [Nguyen et al., 2013]. Therefore, the MROC of water often has to be computed through modelling.

Hydro-economic modelling [Harou et al., 2009] is the science of determining the variations of the value of water in space and time, in order to inform policy on how to promote efficient management of existing water resources. It links water scarcity with water value, and water demand with water price. It relies on an arc-node representation of the system, where nodes represent sources, sinks (e.g. consumption nodes) and storage points, and arcs are unidirectional links connecting nodes. This representation conveys the directional nature of water flow. There are two ways to assess the MROC of water and its variations across space and time:

1) By computing the value of water as the shadow value returned by an optimization model, i.e., the dual or marginal water value associated with the optimal management strategy at the system (or basin) level.
2) By simulating the system and its current management rules, then evaluating the MROC of water under those rules.

Optimisation approaches are in theory the most rigorous way to proceed, because algorithms return the MROC of water if its allocation is economically efficient [Pulido-Velazquez et al., 2008]. The actual value of water for a given use can be matched against that theoretical quantity: if it is lower, this means that allocating water to that use is inefficient, but if it is higher, this means water has been under-allocated to that use. Optimisation algorithms generally aim at maximising benefits or minimising costs over a given period of time. They return the decisions taken to reach this optimum, but also shadow values which can be used to deduce the MROC. Ideally, they must account for uncertainty in future water availability. Otherwise, they may suffer from so-called "perfect foresight" where the optimisation method anticipates extreme events such as floods or droughts up to several years in advance, and adjusts the management strategies to events that cannot be foreseen by water managers in the real world.

Yet, there are few solutions that can find the MROC of water while taking future hydrological uncertainty into account, especially for large systems. For system containing a few reservoirs at most, algorithms from the dynamic programming family, such as [Kelman et al., 1990] are adequate. Yet for more complex systems, these algorithms suffer from the "curse of dimensionality" which makes required computational resources grow exponentially with the number of variables. A notable exception to that is stochastic dual dynamic programming (SDDP), applied to valuating water in large transboundary basins with numerous reservoirs [Tilmant et al., 2008]. SDDP can track the value of water throughout a complex network and under a wide range of hydrological conditions, so that it can be used for water accounting, the science of understanding the financial flows associated with water flowing in a basin [Tilmant et al., 2015].

Optimisation has drawbacks, since even sophisticated methods like SDDP come with assumption that are often far from the groundtruth. In particular, these assumptions tend to not accommodate many of the rules and regulations that play a role in allocating water within a basin. Simulation models however, are meant to represent these. The MROC of can be computed in a simulation model by adding a unit of storage to a given reservoir at a given point in time. These MROC can then be used to derive new rules under the form of pricing strategies, and it has been showed that basin-wide benefits derived in this way could be near-optimal [Pulido-Velazquez et al., 2013].

Simulation and optimization do not live in separate worlds. For instance, water value tables that are deduced from optimization can be used as a rule to balance present and future water scarcity in efficient reservoir operations. This approach, called re-optimisation [Tejada-Guibert et al., 1993] uses functions describing the MROC of water as rules to simulate the system. Conversely, optimization can be used locally, instead of globally (i.e. at the river basin scale), in order to simulate rational decision making from stakeholders. For instance, [Yang et al., 2009] use optimization to simulate the decision of agents depending on their self- and other-regarding preferences, in a way that then allows for computing the MROC of water among other indicators describing the efficiency of the resulting allocation mechanism.

2.2.4 Potential for establishing pricing schemes

Since the MROC increases when water becomes scarce, using rising water prices as a signal of this scarcity is an appealing way of promoting efficient water use [Griffin, 2006]. In fact, recent studies have used values of MROC derived either through simulation [Pulido-Velazquez et al., 2015] or through simulation [Macian-Sorribes et al., 2015] to design scarcity pricing policies. In each reservoir, storage ranges corresponding to a given MROC of water are identified. They define levels of scarcity, and how water should be priced for each scarcity level. None of these studies considers urban use, but evaluating the MROC of water can be done regardless of the nature of the users. The main difference is that the price response may be different for users that need water to produce economic benefits – e.g., farmers – and residential consumers that have discretionary and non-discretionary uses for water. Application of this methodology to the case of residential users will be explored through the Valencia case-study in Section 5.

This type of approach has been backed by regulation in Europe with the Water Framework Directive [European Commission, 2000], which promotes the inclusion of environmental and resource costs in the calculation of recovery costs for water services. The Directive regards water pricing not only as a financial instrument for cost recovery, but also as an economic instrument to regulate demand and create incentives for an efficient water use. For instance, [Riegels et al., 2013] explore how a combination of volumetric pricing of surface water and dynamic pricing of groundwater through varying energy prices can promote water conservation and fulfil the objectives of the Directive. Scarcity-induced price increases signal to water users that, due to supply-demand conditions, water has become more valuable, and that unabated consumption would impose an opportunity cost on other users now, or even on themselves in the future.

Further, setting a water price is a prerequisite to the design of any market or institutional framework wherein water could be traded between different entities that participate. For that reason, establishing pricing schemes is linked with implementing water markets, or institutional mechanisms that mimic them. If they can be enforced, a price tag can be put on all water uses according to its scarcity, which then makes pricing schemes straightforward to envision. For instance, the arc-node formulation typical of hydro-economic models can be used to represent market transactions and even simulate these transactions in a perfect market if it is associated with an optimisation method [Erfani et al., 2013]. Market simulation can then be used to inform the design of new regulations regarding restrictions on water use when it is scarce [Erfani et al., 2015]. Indeed, mimicking an efficient market behaviour enables to devise regulations whose outcomes are as close as possible from those of a perfectly functioning market. Recently, optimisation of the large scale Blue Nile basin through SDDP also led to propose water-sharing mechanisms among riparian countries that are based on the value of the MROC [Arjoon et al., 2016]. One can also find instances where a market is simulated to find the MROC across a river basin. Thus, [Yang et al., 2012] imagined a negotiation mechanism among stakeholders in the Yellow River basin in China that led to deriving the efficient MROC in every sub-region of this large and water-scarce basin.

3. Scarcity pricing

This section builds on Section 3 from Deliverable D5.2, which introduced the dynamic scarcity pricing considered in the SmartH2O project. First, Section 3.1 recalls economic notions about scarcity pricing that were introduced in D5.2, while focusing on the elements that will be interesting in the current deliverable. Then Section 3.2 builds on that to propose a framework for setting the price in an efficient manner in scarcity situations, including details on tariff design and what it means for total payment from consumers to utilities (Section 3.2.3).

3.1 An aggregate representation of urban users' price response

3.1.1 Defining the price response: price elasticity of demand

This paragraph is a reminder about the notion of price elasticity of demand, defined in D5.2. Price elasticity of demand is a quotient which compares the relative proportions by which demand D varies when price p varies. It is generally negative, since demand commonly decreases when prices increase:

$$E(p) = \frac{dD/_D}{dp/_p}$$

The relative change in water consumption is low compared with the relative change in price. This is why residential demand is said to be price inelastic, with values of $E(p)$ typically ranging between -1 and 0. This is a common observation in the residential water pricing literature [Espey et al., 1997; Dalhuisen et al., 2003]. Demand reduction in response to a price increase is dependent on the time elapsed since the tariff change, and often increases as time passes [Dalhuisen et al., 2000; Arbués et al., 2004], but not always [Inman and Jeffrey, 2006]. This unpredictability is the reason time dependence is kept implicit in the above equations. Moreover, smart metering can help including pricing in comprehensive strategies that manage demand through a combination of customer engagement, awareness campaigns, detailed personalized feedback on consumptive behavior, gamification, etc. Tariff changes are expected to be most effective then [Hill and Symmonds, 2011], but impacts on price elasticity have yet to be investigated. The development of smart metering takes place at a time when new avenues for engaging the public, and modeling their behaviors, are being explored [Fraternali et al., 2012]. In particular, user modeling is seen as a promising tool to help designing personalized water demand management strategies [Cominola et al., 2015].

3.1.2 The demand curve

The demand change that is the consequence of a price change is obtained by integration of the price elasticity of demand function from an initial price p_0 to new price p':

$$\frac{D(p')}{D(p_0)} = \exp\left(\int_{p_0}^{p'} E(p)\frac{dp}{p}\right)$$

Therefore, knowing the price elasticity of demand is enough to understand the relationship between pricing and consumption. This relationship is graphically described by the notion of demand curve [James and Lee, 1971] (Figure 1), which tells how much water residential users would consume depending on how much they have to pay for an additional unit of water.

A major advantage of the demand curve is that it can be interpreted graphically [Harou et al., 2009]. Indeed, the area below the curve, between the y-axis and the current demand level, is the gross benefit from water consumption. It can be decomposed into two areas, (A) and (B). Area (A) is the rectangle defined by the x- and y- axes and by the dotted lines. It represents the market value of water, that is, the amount paid by residential customers to get that water (assuming it has been delivered at a uniform price). Then, area (B), the other part of the gross benefits, is called the consumer's surplus. It can be interpreted as the net benefit of residential users from consuming water.

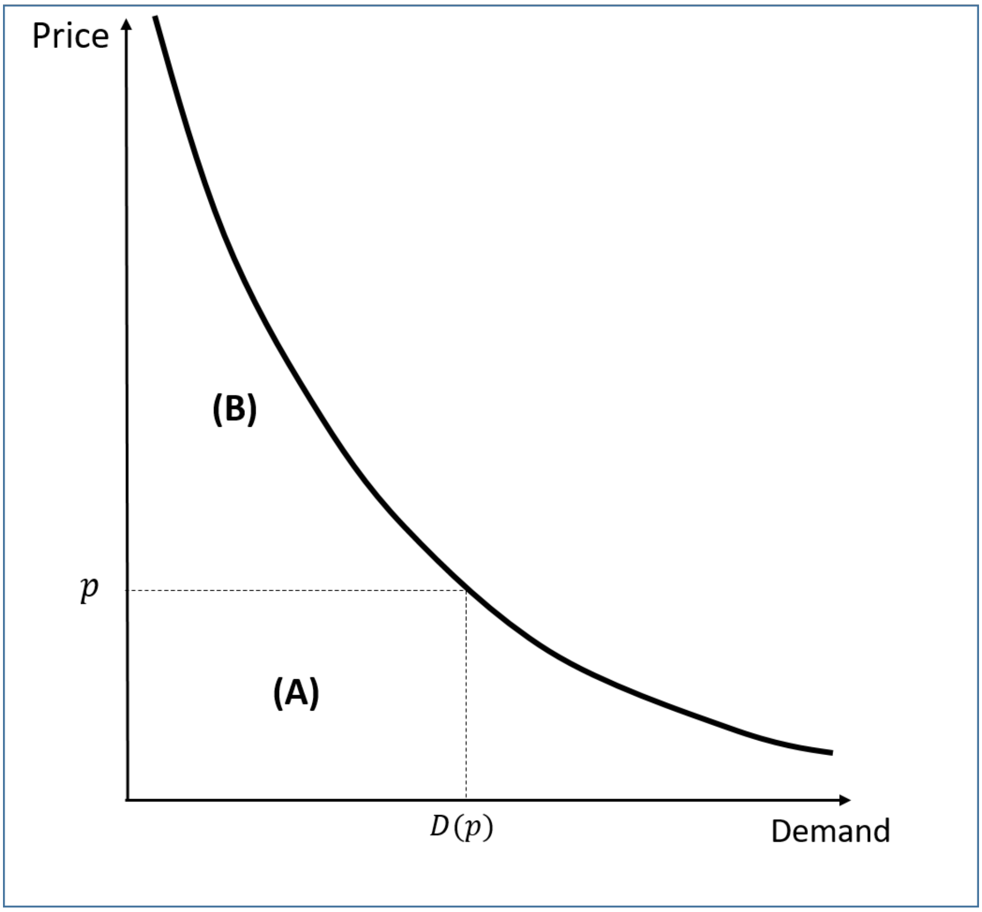


Figure 1. Demand curve with (A) market value and (B) consumers' surplus.

For this study, the 'point expansion' method [James and Lee, 1971] is used for extrapolation from a single point on the demand curve. This technique is easy to apply since it can be used to obtain an entire demand function estimate from a single value of quantity and price, and a single estimate of the price elasticity of demand. The latter is the normalised derivative at the point defined by the two formers. These inputs are commonly available.

In the point expansion method, one must choose which parameters thus defined at a point will be assumed to be constant. On one hand, if the demand curve is assumed linear, the slope given by the elasticity at the point is taken as a constant. On the other hand, elasticity itself can be taken in a constant, yielding a demand curve shaped like an exponential, similar to Figure 1.

The disadvantage of this method is potential oversimplification of the demand curve. Water demand may not exhibit linearity or constant elasticity across the full range demands and prices. Therefore, these two functional forms may not correspond to actual human behaviour in situations far from the point of expansion. Yet, this method can be potentially used for all sectors of water demand (e.g., residential, commercial, recreation, hydropower, etc.), [Griffin, 2001].

3.1.3 Response to a demand reduction tariff

Demand reduction may take place over any arbitrary period of time. Price must be raised from base price p_0 to demand reduction price p_r in order to achieve a relative change (reduction) X_r in the demand *D*, e.g., -5% or -10%:

$$\frac{D(p_r)}{D(p_0)} = 1 + X_r$$

Combined with the relationship between demand and elasticity at the beginning of Section 3.1.2, the latter yields the relationship between price p_r, target demand reduction $-X_r$, and elasticity *E(p)*:

$$\exp\left(\int_{p_0}^{p_r} E(p)\frac{dp}{p}\right) = 1 + X_r$$

This equation enables the computation of the demand reduction price p_r if function *E(p)* is known – or if the demand curve is derived through the point expansion method. Figure 2 provides information on the financial implications of raising prices to reduce demand. Reducing demand incurs a revenue loss, because there is a less water to make a revenue from (blue hatched rectangle). It also leads to a revenue gain, because remaining water is sold at a higher price. Since residential water demand is price inelastic with elasticities noticeably smaller than -1 (in absolute value), the revenue gain generally exceeds the revenue loss.

To make the tariff revenue neutral, the revenue gain and the revenue loss must be equal (see Figure 2). Otherwise, there is a revenue gain for utilities, and therefore a financial loss to customers. One must observe that to ensure revenue neutrality while enforcing demand reduction, it is sufficient for the marginal value of residential water to be at p_r. Revenue neutrality can then be achieved by designing a scheme whereby utilities forsake the excess revenue (black rectangle; more details on this in Section 3.2.3).

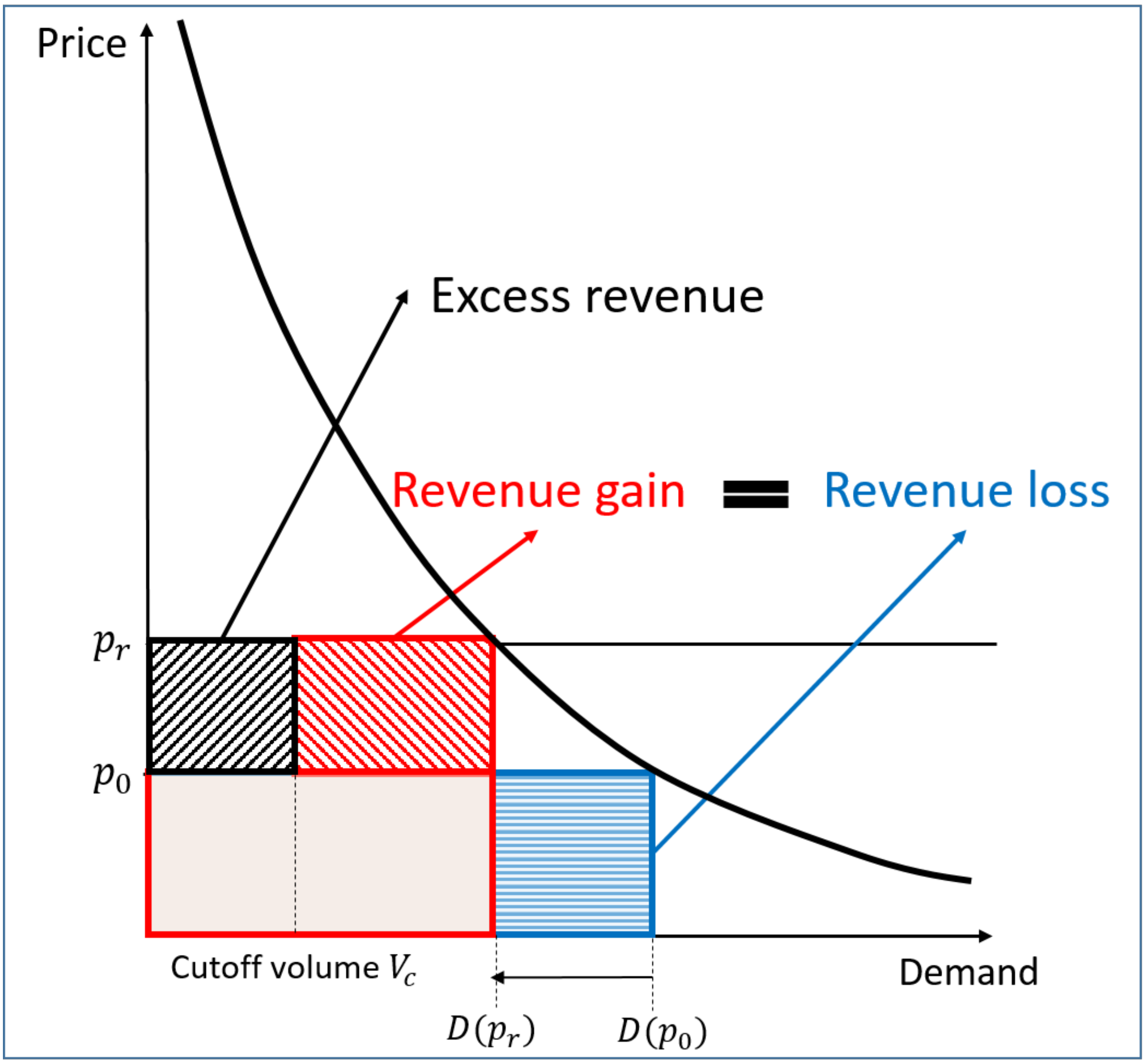


Figure 2. Raising prices to manage demand, and financial implications.

The cutoff volume $V_c(p_r)$ separating prices p_0 and p_r has a dual interpretation regardless of the details of the tariff used. It is at the same time the maximal volume that can be charged at the base rate p_0, and $(D(p_r) - V_c(p_r))$ is the maximal volume that can be charged at rate p_r while ensuring revenue neutrality. The cutoff volume is computed by noting that the red and blue hatched areas must be equal:

$$(p_r - p_0).\left(D(p_r) - V_c(p_r)\right) = p_0.\left(D(p_0) - D(p_r)\right)$$

3.2 Efficient scarcity pricing

3.2.1 Demand curve and water value

Efficient pricing considers the value of water as a commodity, and does not take into account the costs that usually justify the billing of water to residential consumers, pre-consumption treatment costs, water delivery costs, post-consumption treatment costs. The demand curves introduced in Section 3.1 consider the total cost incurred by residential consumer. That cost is the sum of the cost of water as a commodity or “raw” water, and of the treatment and delivery costs. Therefore, one can deduce the residential demand for “raw” water from the “tap-delivered” demand curve by subtracting treatment and delivery costs from the total cost (Figure 3).

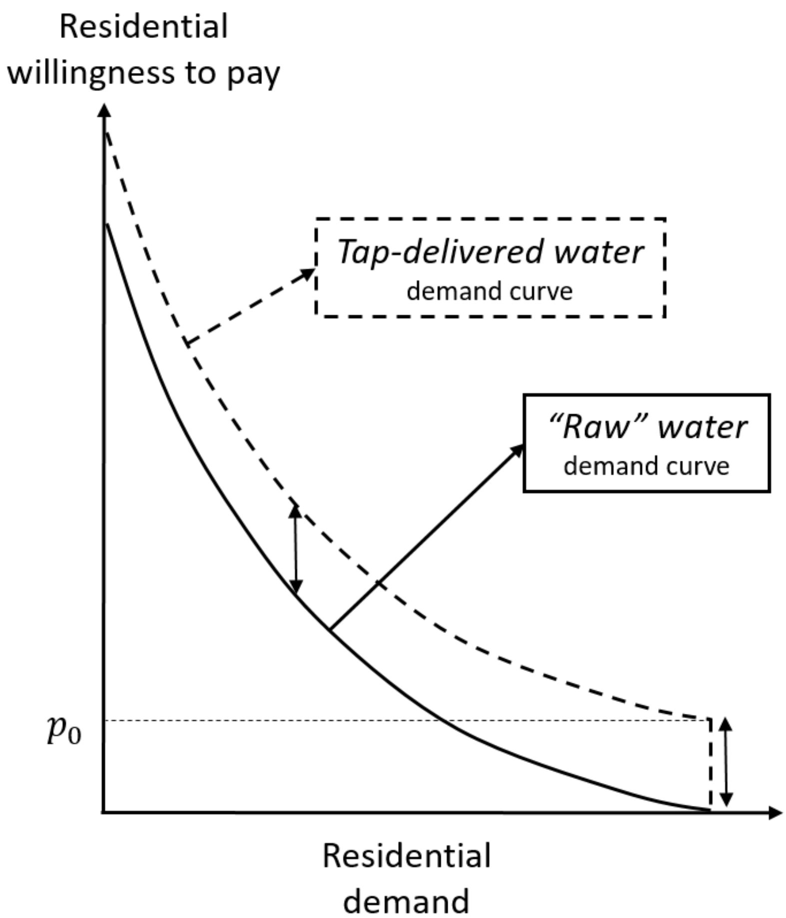


Figure 3. Relationship between the residential demand curves for "raw" and tap-delivered water.

Assuming that these costs do not change with water scarcity, and that non-scarcity or baseline price considers “raw” water to be free, one can easily deduce the demand curve for “raw” water from the tap-delivered one. These assumptions will be used in both case-studies (Sections 4 and 5) to facilitate the computation of pricing policies in scarcity situations.

In reality, one can expect that the cost of treating and delivering water could increase when water becomes scarce, as utilities turn to water sources that are less easy to access or to use. For instance, it is generally more expensive to make salt water fit for consumption through desalination, than it is to treat ordinary river water. Yet, utilities include these costs in their static, year-round residential prices, therefore the water prices used in the case-studies recover costs in the same way regardless of the scarcity situation. What is more, the relevant figures for this deliverable would be concerned with how much treatment costs increase with scarcity. For instance, we would need to know the price difference between desalinated water and other sources. These are figures that evolve quickly with technology, so we would need the utilities' own estimates, and utilities regard such data as confidential. In this deliverable, the difference between the "raw" and the "tap-delivered" demand curve is assumed to be constant, except in Section 4.3.3.

3.2.2 Setting the price according to water scarcity

Residential scarcity pricing defines efficient prices according to the cross-sectoral value of water, that is, the marginal water value from all other uses – agricultural, industrial, environmental, etc – at the abstraction point. The efficient allocation of water among competing sectors is defined by the equimarginality principle, which states that the net benefit function is maximized when the net marginal benefits per unit of water are equal in all use sectors. For the case of two sectors, or when an efficient cross-sectoral price of water already exists for all non-residential uses, the equimarginality principle can be illustrated graphically [Young, 1996] by representing on Figure 4 and Figure 5 the demand curves for residential (from upper left to lower bottom) and for other uses (from the right-hand axis).

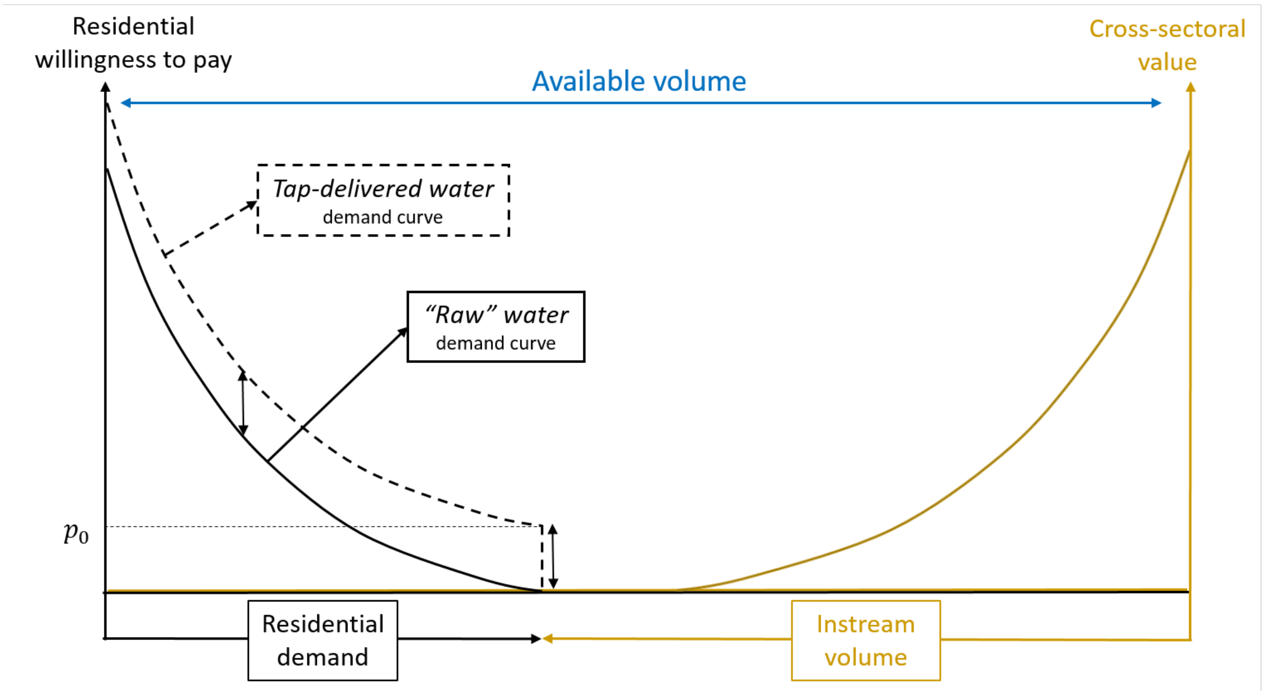


Figure 4. Enough water is available: no scarcity cost

In a non-scarcity situation (Figure 4), residential water is delivered at its base rate p_0. Then, water itself has no value, and the base volumetric rate p_0 is typically a reflection of the utility's average costs in the common case where prices equal average cost. When there is water scarcity (Figure 5), the two curves are crossing, and the optimal allocation corresponds to the price given by their intersection. This price π represents the marginal economic value of water as a resource, also called its shadow value. Water price at the tap is then given by:

$$p_r = \pi + p_0$$

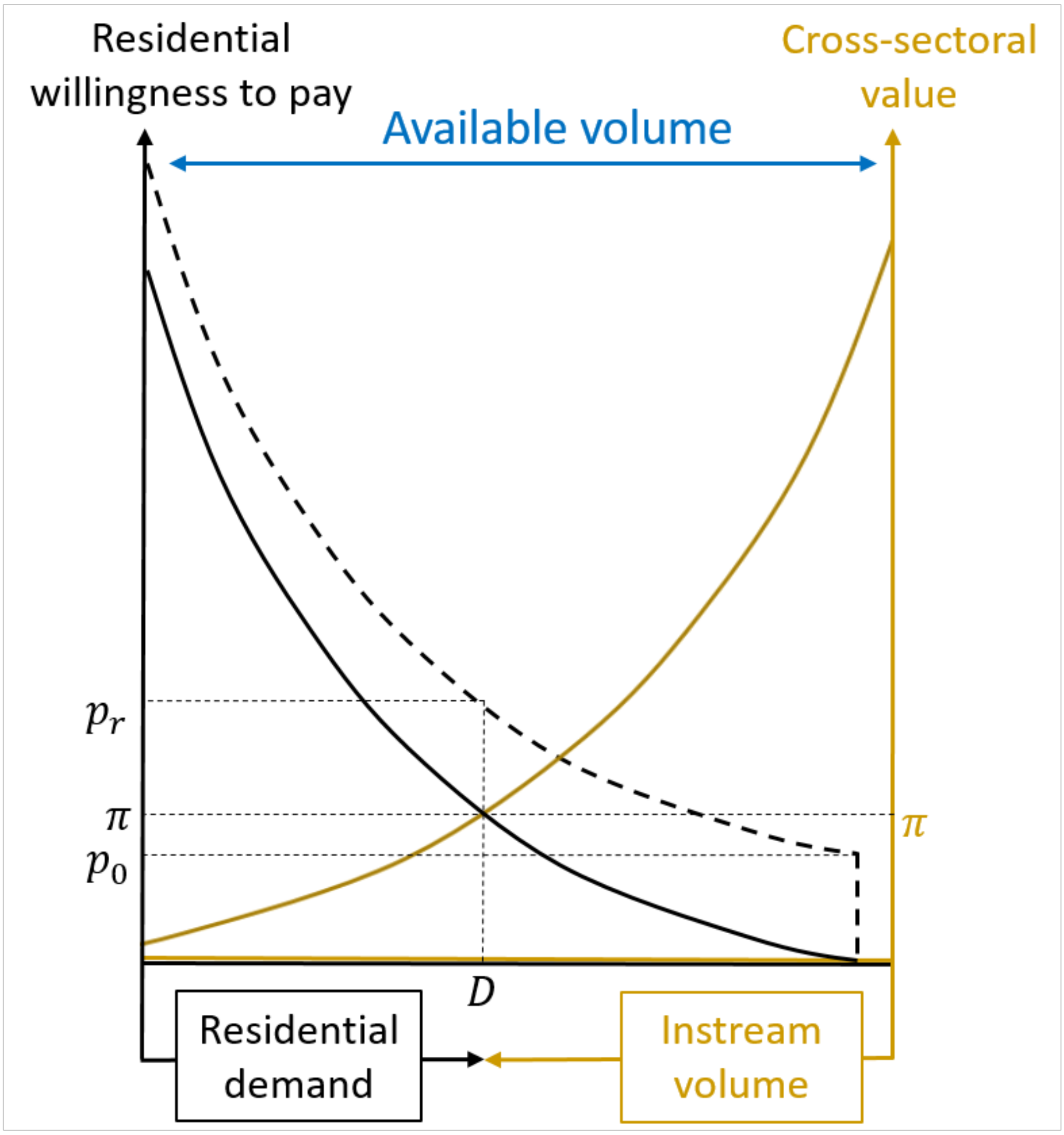


Figure 5. Efficient water pricing in a scarcity situation

3.2.3 Possible uses of the excess revenue raised through scarcity pricing

Scarcity pricing is a demand reduction tariff, expected to produce excess revenue for utilities because water is an inelastic good. This raises an acceptability issue, as surplus funds may be viewed as a new environmental tax. In the UK, revenue neutrality is a regulatory imperative enforced by Ofwat, the government body that oversees the functioning of the private water market [Ofwat, 2009]. Yet, mechanisms that reduce demands without increasing prices lead to deficits for water utilities [Hill and Symmonds, 2011].

Revenue neutrality can be achieved by designing a scheme whereby utilities forsake the excess revenue. For instance, any form of increasing block tariff (IBT; see e.g. [Griffin, 2006]) achieves this. IBT can be designed with multiple objectives, sparking debate regarding matters such as equity among users. For instance, even the simplest, two-block IBTs are deemed more equitable than uniform pricing at price p_r [Ward and Pulido-Velazquez, 2009]. However, recent studies cast doubt on the narrative that the poor benefit from IBTs, in part because the assumed correlation between per-capita income and household consumption does not necessarily reflect reality [Whittington et al., 2015], and a uniform volumetric charge with rebates is superior in some cases [Boland and Whittington, 2000].

Further, scarcity pricing in itself may be criticized on the ground of distributional fairness. For instance, low-income households are more price elastic, and could bear a worryingly large burden in terms of consumption reduction [Olmstead and Stavins, 2009]. Public acceptability of scarcity pricing could be enhanced through rebates and/or earmarking policies. Rebates schemes are a way to meet the revenue neutrality constraint, and can also be included in water bills to address equity concerns, i.e. larger shares of surplus funds can be returned to lower-income households. However, in order to preserve the incentive power of scarcity pricing, rebates have to be lump-sum transfers, i.e. independent of consumption [Olmstead and Stavins, 2009; Mansur and Olmstead, 2012]. These transfers can also be implemented through so-called “social tariffs” designed at guaranteeing access to water to the most vulnerable segments of a population. Secondly, a part of additional revenues from scarcity pricing can be earmarked for water conservation investments and other environmental quality improvement. Earmarking revenues to activities that produce environmental benefits increases the popularity of environmental regulation measures [Kallbekken and Aasen, 2010].

4. London case-study

This section presents the supply-demand modelling and its results on scarcity pricing from the London case-study. The London context is presented in Section 4.1, showing that water scarcity in the area is due to a strong demographic growth which ultimately puts residential consumptions in direct competition with riverine ecosystems when there is a drought. Then, Section 4.2 presents the hydro-economic modelling leading to the evaluation of scarcity pricing in that context. Finally, Section 4.3 presents the results of this studies and discusses the potential benefits and limitations of scarcity pricing.

4.1 Context and objectives

4.1.1 Presentation and context

The Thames river is 346 km long and flows eastward through Southeast England and into the North Sea. It supplies water to some of the most densely populated areas in the UK, and currently provides about two thirds of the water supply of the Greater London – the rest is provided by groundwater [Environment Agency, 2009]. The Greater London is an administrative entity of over 8.5 million (M) inhabitants, at the core of a metropolitan area topping 13M inhabitants. Population in that area is growing, fuelling concerns about future water supplies in an area that is already classified as water stressed according to UK standards [Environment Agency, 2007]. These concerns have motivated Thames Water, the utility that serves most of the Greater London, to investigate means of closing the expected gap between supply and demand through a varied portfolio of future water supply options (Figure 6). They are also looking to act on the demand side of the equation, and that provided part of the motivation for launching a 15-year smart metering roll-out set to equip a sizable proportion of the 3.3M households they serve [Rasekh et al., 2016].

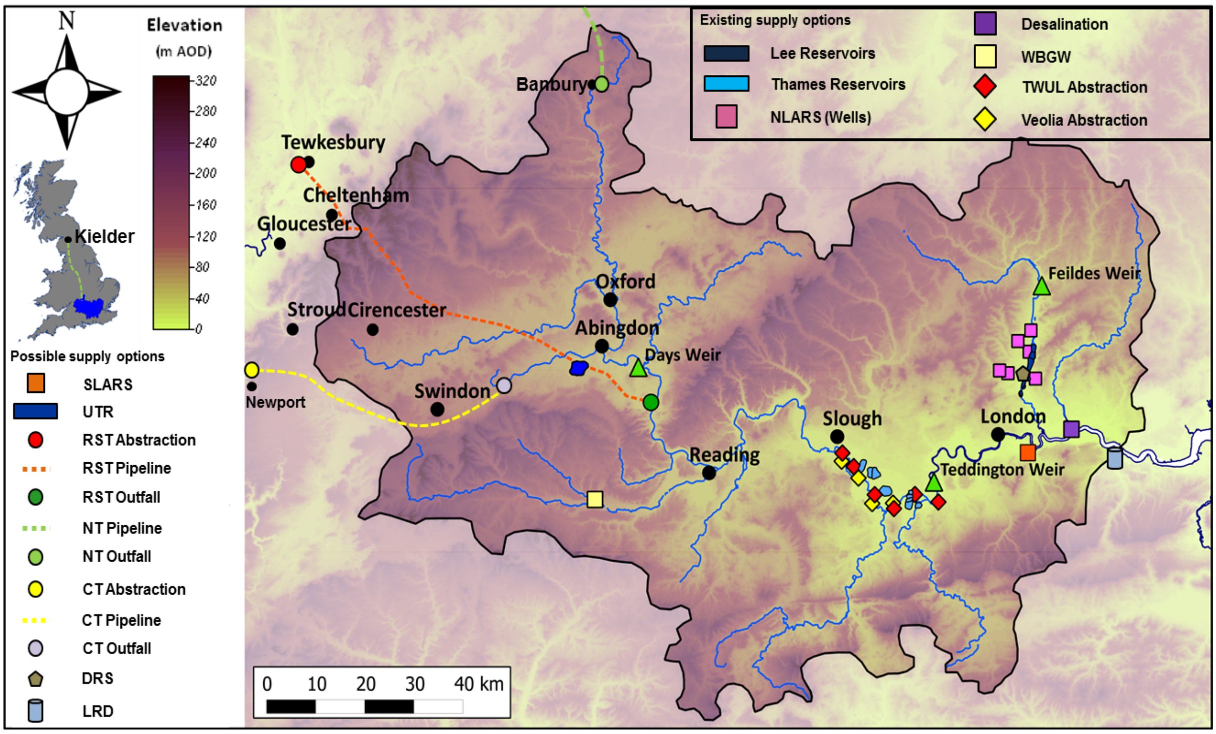


Figure 6. Thames river basin, with abstraction points and present and future water supply options.

Urban uses are the main source of anthropic water consumption in the basin, but other sectors benefit from the water flowing in the Thames. Biodiversity conservation is at stake, but the services it provides do not stop there. Navigation and tourism are activities that benefit from the presence of the river. Riverfront property valuation depends on how scenic and ecologically healthy the river is. The presence of free-flowing water in the Thames that benefits all these sectors simultaneously, and this is what is meant in the remainder of this section through the generic phrase of “environmental flows”.

4.1.2 Objectives

Owing to the expected demand growth, water scarcity is much more of a future challenge than a current one. Scarcity pricing also is a future possibility, seeing that the smart meter rollout is due to continue until 2030, and that smart metering is a precondition to scarcity pricing. Most households are currently unmetered [Thames Water, 2014] so it is anticipated that it may take a few years for residential users to get used to the idea of a volumetric charge for water, before any dynamic pricing or incentive scheme can be envisioned.

For these reasons, studying scarcity pricing in London is a forward-looking exercise. Therefore, the goal of the study is **to examine the potential of scarcity pricing depending on a series of key uncertain factors**.

The potential of scarcity pricing will be assessed through:

- Impact on environmental flows, comparing with results from a rule-based simulator.
- Impact on residential water prices.

The uncertain factors studied will be:

- The value of environmental flows, on which the assessment of the MROC hinges. The higher it is, the higher the MROC from diverting water away from the river.
- The demand response, depending on the value of the price elasticity of demand, but also on the point expansion method used to derive the demand curve (constant elasticity vs. demand slope).
- Policy implications of sending consumers a signal when more expensive supply sources have to activated (treatment and delivery costs are considered dynamic). An example will be taken, considering a desalination plant whose activation triggers a price increase (Section 4.3.3).

In this forward-looking exercise, the details of the possible tariffs will not be studied. Besides, demand growth is expected to play a much larger role than climate change in altering the reliability of urban water supply by mid-century [WRMP, 2014]. Therefore, this study is to use historical time series spanning 85 years (1920-2004) in order to focus on the impact of the increased demand on the supply-demand balance.

4.2 Methodology: hydro-economic modelling

This section first presents the simulation model used, with a focus on the features of the system that are of most interest for this study (Section 4.2.1). Then, the method for computing the MROC of water, basis for designing dynamic pricing schemes, is explained (Section 4.2.2). This computation assumes that the demand curves for both environmental flows and urban consumption are known, which is not the case, so the assumptions between the construction of these curves will be discussed in Sections 0 and 4.2.4. Finally, Section 4.2.5 will propose three scenarios to be studied in the result section (4.3).

4.2.1 Supply-demand model of London

This section uses the IRAS-2010 model [Matrosov et al., 2011] for the Thames Valley and the Greater London (Figure 7). This is a rule-based simulation model that allocates water according to the regulations and agreements in force in the basin. It uses historical flows from 1920-2004 with a weekly time step, and combines them with projected demands for 2050 to identify the system’s possible non climate-change related vulnerabilities while investigating the impact of various water supply options – e.g. desalination – and of demand management strategies such as water use restrictions.

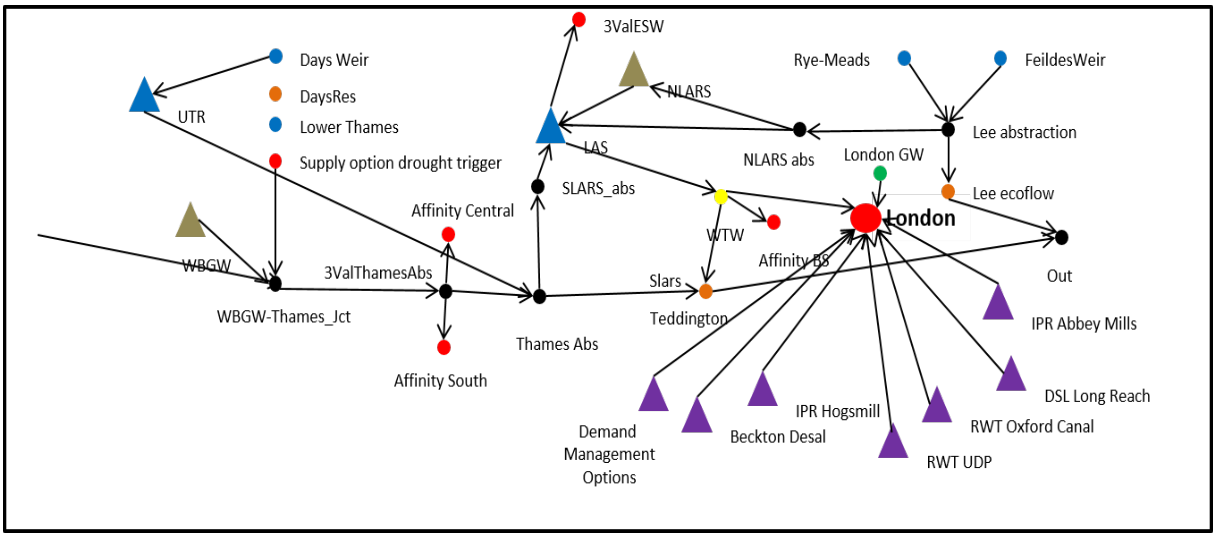


Figure 7. Schematic of the London water supply-demand system.

Components of this network are bound together by a number of rules and regulations, and these are likely to increase as an increased demand makes water more scarce, pushing stakeholders to secure the necessary supplies through regulations. In this complex network, this study focuses on how scarcity pricing could be used to re-equilibrate demand for urban consumption and environmental flow downstream of the water intakes. Therefore, the focus is on the role of storage in the London Aggregated Storage (LAS) for those two uses (Figure 8), and on the interplay with other supply sources in London.

In the model all the London reservoirs are aggregated into one reservoir with total capacity of 202 828 Ml – about three months of water demand. In the WARMS model they are lumped into three: north Thames reservoirs, south Thames reservoirs and Lee Valley reservoirs. In fact, the Thames and Lee storage is connected by the Thames-Lee tunnel with a capacity of 410 Ml/d (flow goes only in 1 direction: Thames-Lee). Splitting the reservoir in the modelling would introduce considerable extra complication.

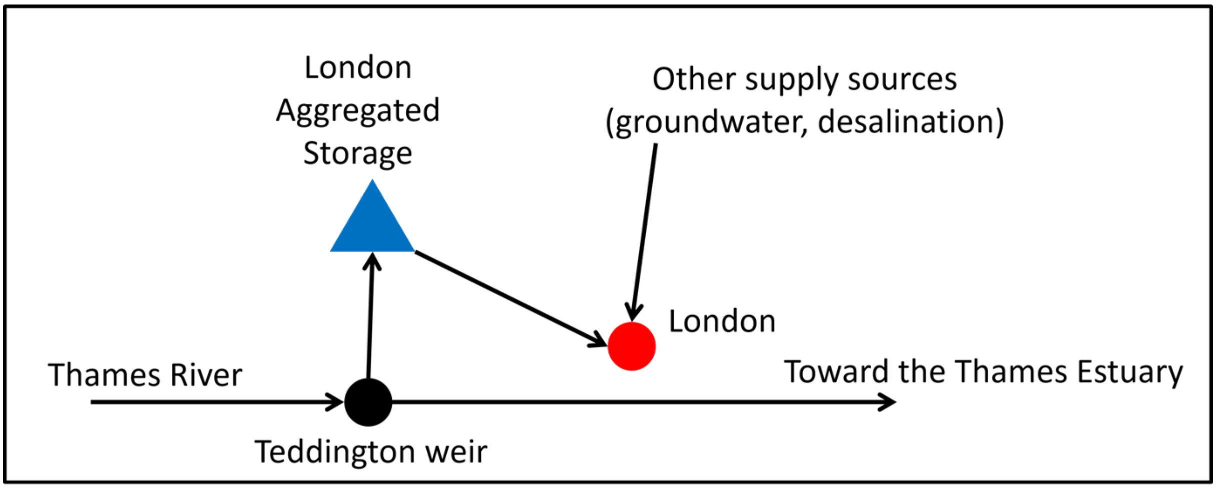


Figure 8. Detail of Figure 7: the LAS, London demand node, and environmental flows through London.

Groundwater, in the representation, is considered as supplying demand directly rather than adding to storage. Demand calls on groundwater first and is always considerably more than it can supply. London demand is modelled as the expected demand in 2050 as given in the Thames Water WRMP14 [Thames Water, 2014]. A monthly profile is used (Table 4).

Month	1	2	3	4	5	6	7	8	9	10	11	12

Factor	0.989	0.977	0.965	0.964	1.006	1.014	1.043	1.056	1.001	0.975	0.99	1.02

Table 4. Monthly factors for London's demand

The environmental flow normally required at Teddington Weir is 800 Ml/d but can be reduced in increments according to the LTCD. As for the Gateway desalination plant, this node is modelled as an infinite reservoir that releases water straight into London when the Thames flow at Teddington goes below 3,000Ml/d.

4.2.2 Computing the MROC of water

Restrictions are enforced when London's aggregated storage (LAS) in the Thames basin's reservoirs drops below given levels; these levels vary according to the time of year. The planning of these restrictions is called the Lower Thames Control Diagram (LTCD; Figure 9). These restrictions impact two sectors.

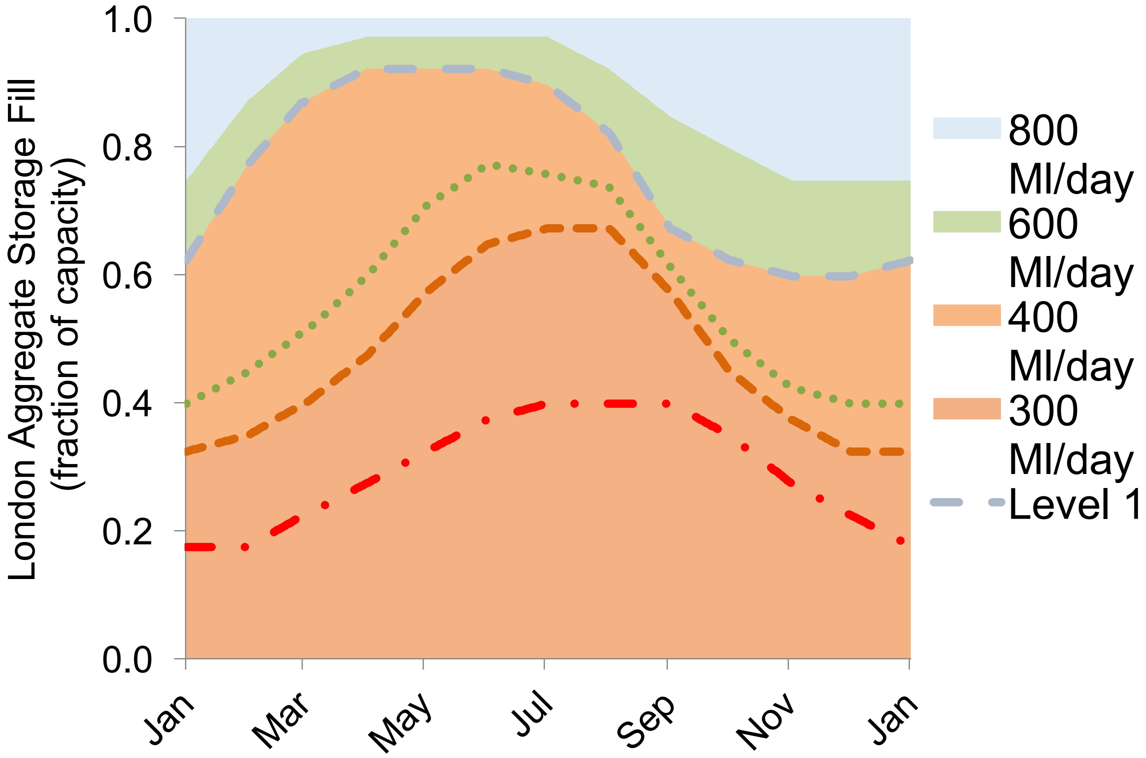


Figure 9. The Lower Thames Control Diagram (LTCD).

On one hand, the London demand is reduced when LTCD restrictions are invoked. Demand reductions for each successive LTCD level are assumed to be achieved through a series of demand management strategies of increasing severity. Four levels of demand management targets exist. Dynamic scarcity pricing is not considered among these demand management measures.

On the other hand, the Thames river flow at London is regulated at Teddington weir, with a minimum requirement of 800ML/day to reflect river flow benefits such as navigation, recreational and environmental values. If storage in the LAS drops below a certain level, this requirement is not fully met, implying losses to these sectors. The numerical values on Figure 9 show the amplitude of these restrictions (colour zones). Therefore, environmental allocations drop below the minimum requirement in the case of a drought.

This study aims at reallocating water between urban consumption and the environment when restrictions are enforced by using scarcity pricing. The amount of water allocated to both uses is the sum of urban demand and environmental flows simulated by IRAS. It is reallocated on a week-by-week basis in an economically efficient way, i.e., by equating the MROC of untreated water for both uses. This is similar to Figure 5. Scarcity pricing adds the marginal water value for environmental use to the usual value of water, which only takes treated water into account. In this study to assess the potential of scarcity pricing, it is assumed no other demand management scheme can reduced consumption farther than pricing does. However, one can assume that demand management strategies are employed to improve the price response (by influencing the price elasticity of demand defined in Section 3.1.1).

Computing the MROC of water for the two uses considered here can only be done if demand curves are known or assumed. Deriving these is the topic of the two next paragraphs.

4.2.3 Constructing the demand curve for environmental flows.

Demand curves for environmental flows, which amount to estimating the general population's willingness to pay is for different levels of environmental flows, have not been estimated for London. In this data-scarce context, a simple linear environmental demand curve is used, which is consistent with previous theoretical studies [Yang et al., 2009; Giuliani et al., 2014]. In this case it is sufficient to know the aggregate environmental economic benefits of having river water in the Thames in order to derive the demand curve (Figure 10).

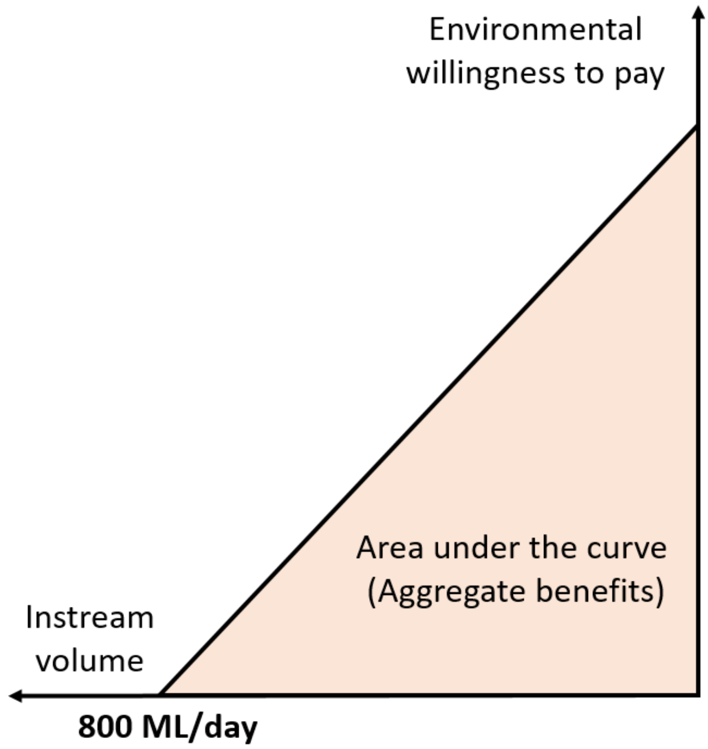


Figure 10. Linear environmental demand curve for the London case-study.

Parameterizing the demand curve is a challenge because there are many ways in which river flows are valuable [Kulshreshtha and Gillies, 1993]. Two willingness to pay studies provide a similar evaluation of the environmental value of Thames River flows [Thames Water, 2005; Eftec, 2015]. Both are based on stated preferences from respondents in the Thames Water region in the context of the construction of the Thames tideway tunnel, a large new infrastructure aimed at eliminating combined sewers overflow. These studies both report an aggregate annual value of around £250M that encompasses a series of ecosystem services brought by the river thanks to this infrastructure. Specific ecosystem services used by a fraction of the population add to this total, but willingness to pay studies report comparatively much smaller value for these. For instance, [Peirson et al., 2001] evaluated that taken together, anglers evaluated the value of angling to be £12M. This annual aggregate value of £250M is a lower bound for the value of the Thames' water. It can therefore be used as a baseline value of environmental flows, but the valuation should not stop there.

Indeed, the value for flows in London's Thames River goes beyond ecosystem services and associated recreational benefits. For instance, riverfront location bolsters the value of both new and existing real-estate developments (Cassidy, 2013). The river contributes both directly – cruises, touristic attractions, riverfront venues – and indirectly – through its place in popular culture – to tourism revenues, estimated at £15 billion a year from overnight visitors and up to £26 billion a year when accounting for day trips to London (Visit England, 2016). Given the amounts at stake, even a minor contribution of a few percentage points to the value of riverfront development and to the revenues of tourism might represent several hundred £M. For instance, a percent decrease in tourism benefits is worth between £150M and £250M.

Therefore, the baseline value of £250M per year is a lower bound fraught with the following limitations:

- It only accounts for part of the ecosystem services brought by the presence of river water from the Thames River.
- It does not account for the impacts on the tourism industry and on property valuations, each of which could be on the order of a few hundred £M.

To investigate the possible implications for scarcity pricing, total values of instream flows worth £500M per year ('medium value scenario') and £750M per year ('high value scenario') are compared to the baseline estimate of £250M per year. These values are then disaggregated at the weekly time step, and put into weekly demand curves such as the one represented on Figure 10.

4.2.4 Representation of the demand response

Since many households in Southeast England were not metered until recently, there is not much data available from which to derive the demand response to price signals. What is more, for the same price the price elasticity of demand can be influenced by a variety of parameters – as discussed in Deliverable 5.4. In fact, representing the demand response depends 1) on the choice of a value for the price elasticity of demand, and 2) on the method used to derive the demand curve from that value.

Deliverable D5.2 used a meta-analysis of price elasticity studies to understand the factors that influence the price elasticity of demand thanks to a statistical regression analysis. It then used this analysis to provide estimates of the price elasticity of demand for the project's case-studies, including London. It found value of $E = -0.3$ and $E = -0.4$, depending on the details of the tariff used. These two values are considered in this study, alongside a third estimate $E = -0.5$ which represents the idea that dynamic scarcity pricing can be used in conjunction with other demand management tools in order to enhance the price response.

Two methods for deriving the demand curves are considered here. They are the two variants of the point expansion method described in Section 3.1.2. On one hand, considering a constant price elasticity of demand, an exponential demand curve is derived. It conveys the idea that it becomes very hard to reduce water consumption once the discretionary uses have been eliminated. Therefore, the willingness to pay for the first cubic meter every month, for instance, is high. On the other hand, using the point expansion method with a linear demand curve seems logical in a case where the environmental demand curve is also linear. In both cases, the demand curve is derived from the estimate of E used, and from one known point of the demand curve. The projected 2050 demands for London presented in Thames Water's 2014 Water Resource Management Plan [Thames Water, 2014], are used alongside the current volumetric price of £2.05 per cubic meter used by Thames Water (£1.26 per cubic meter for water, and £0.79 per cubic meter for groundwater).

4.2.5 Demand response scenarios

The three scenarios considered are as follows:

1. Scenario 1: demand curve with constant price elasticity of demand
2. Scenario 2: linear demand curve (the same price change leads to the same change in consumption, in non-relative terms)
3. Scenario 3: the cost to deliver water varies dynamically: when the desalination plant is triggered, water prices increase to reflect the fact that desalinated water is more expensive to obtain.

For each scenario, nine cases are considered, corresponding to the three values of the environmental benefits defined in Section (£250M, £500M and £750M) and to the three values of the price elasticity of demand defined in Section (-0.3, -0.4, -0.5).

4.3 Results

The three demand response scenarios presented in Section 4.2.5 are presented successively. For each scenario, indicators associated with protection of environmental flows and with price increases associated to dynamic pricing are presented. These indicators are:

For environmental flows:

- Mean violation during the weeks when there is a violation (flows under 800 ML/day) in the rule-based IRAS simulation. When there is no violation, there is enough water being allocated, and no need for scarcity pricing. In the rule-based simulations, there are 12.9 weeks per year with such restrictions, indicating that water scarcity could be a chronic feature of the Thames basin's future. That corresponds to almost 25% of the time.
- Number of flow events during which flow went below 400 ML/day (severe scarcity) during the 85 years period, and duration of these events.
- Number of flow events during which flow went below 200 ML/day (extreme scarcity), and duration of these events.

For price increases:

- Frequency of 10%, 50%, 100% and 150% price increases in weeks / year.
- Duration of the price increase events for these same price thresholds.
- Number of recorded events during the 85 years for these same price thresholds.

4.3.1 Scenario 1: residential demand curve with constant elasticity

Scarcity pricing is effective at reducing the average environmental flow deficit (Figure 11) compared with the rule-based simulations that implemented 4 pre-defined levels of water restriction. As expected, this effectiveness increases when the elasticity goes from -0.3 to -0.5, because it takes less of a price increase to achieve the same reduction in consumption. Yet, the effect of this parameter is not as important as expected, as average deficit is only 20% smaller, with -0.5 than with -0.3 when the annual value of environmental flows is £250M; the difference does not exceed 30% for the medium and high values of environmental flows. In contrast, the effect of the evaluation of the benefits from environmental flows is more important, as it almost 40% for all three values of price elasticity of demand. Yet, it is interesting to note that even in the low-elasticity, low-environmental value scenario, the average deficit is reduced by 23%, from 275 ML/day during a week to 212 ML/day during a week.

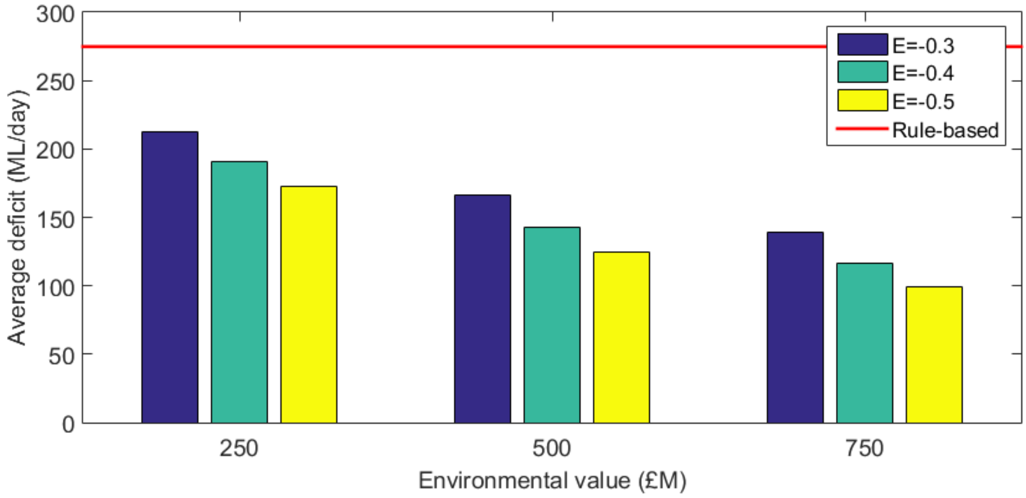


Figure 11. Average deficit for weeks with an environmental flow deficit in the rule-based simulation, scenario 1.

When it comes to assessing the duration and number of occurrence of the severe (less than 400 ML/day during a week) low-flow events during the 85-year period, scarcity pricing still proves very effective compared with the planned water use restrictions (Figure 12). This is mainly due to the fact that the number of events during which this circumstance happens is reduced from 42 in the rule-based simulation to a maximum of 10, reached in the low- and medium-elasticity cases with environmental value of £250M. Again, environmental value seems to have a greater influence than elasticity on the occurrence of these events. When it comes to duration though, average duration is equal or bigger than the rule-based one for most of the low- and medium-elasticity cases. Yet, this can be attributed to the fact that this average is computed over drastically different samples. In reality for each of the severe scarcity events with dynamic pricing, event duration is longer in the rule-based case. This being said, it is noteworthy that high-elasticity, medium- to high- (environmental) value cases lead to noticeably shorter severe scarcity events – around 4 weeks on average.

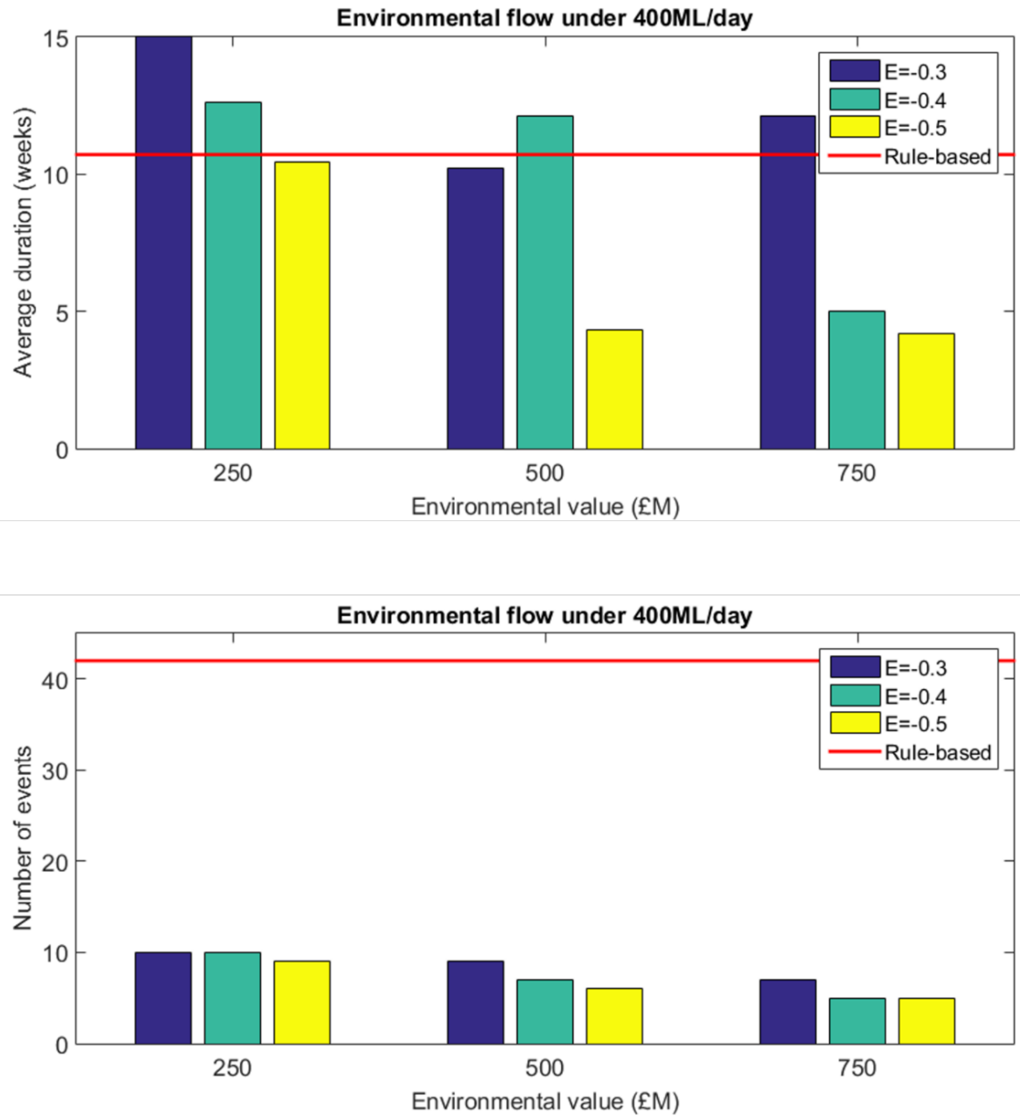
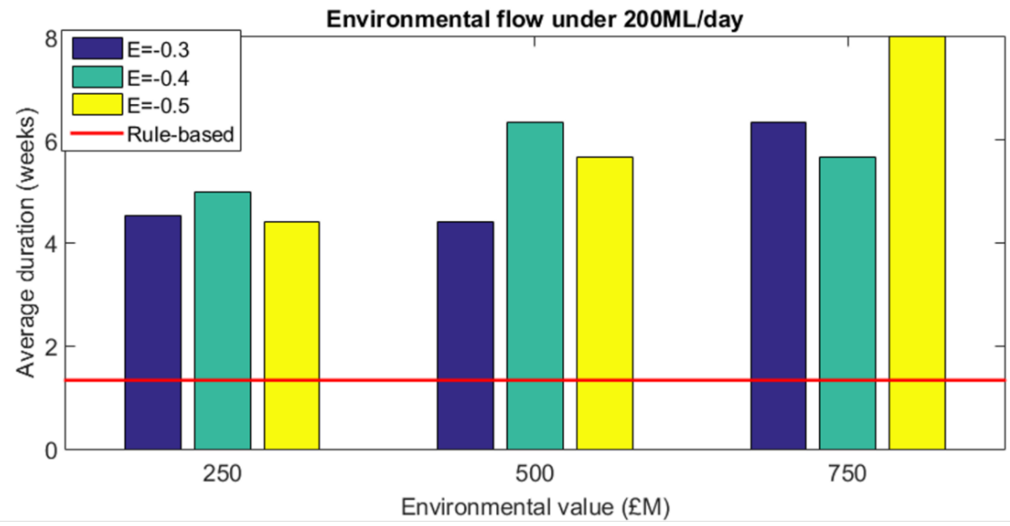


Figure 12. Duration and number of occurrences of weeks with environmental flows below 400 ML/day, scenario 1.

However, in each of the nine cases, half or more of the severe scarcity events are also extreme scarcity events (under 200 ML/day during a week; Figure 13). Only for the high-value, high-elasticity case are there less events with scarcity pricing than in the rule-based simulation, where that remains exceptional – 4 weeks in total, divided in 3 separate events. Yet, these 2 events are long, as they total 16 week of extreme scarcity. In other cases, these events are equally long or longer, and extreme scarcity happens in other occasions (up to 13 times in the low-elasticity, low-value case).

Thus, in scenario 1, scarcity pricing offers less protection to extreme scarcity than the restriction rules. This because when water is very scarce, the constant price elasticity of demand means that the same relative price increase is required to achieve the same relative demand reduction. Prices are already high, and demands are already low, compared to normal, so in fact bigger price increases achieve smaller demand reductions. Then, the residential users are the one being protected by the residential tariff instead of the environment. Indeed, the environmental demand curve is linear, which means the price increases linearly as the deficit increases. In the worst occurrences of drought, this leads to letting the river drying up completely, and this happens for every of the nine cases, from 14 weeks in the high-value, high-elasticity case to 19 weeks in the low-value, low-elasticity case. The shape of the demand function, reflecting the fact that residential demands are not 100% reducible, leads to similar results regardless of other parameters. In contrast, the week with the lowest environmental flow during the 85-year rule-based simulation had a flow of 131 ML/day during a week.



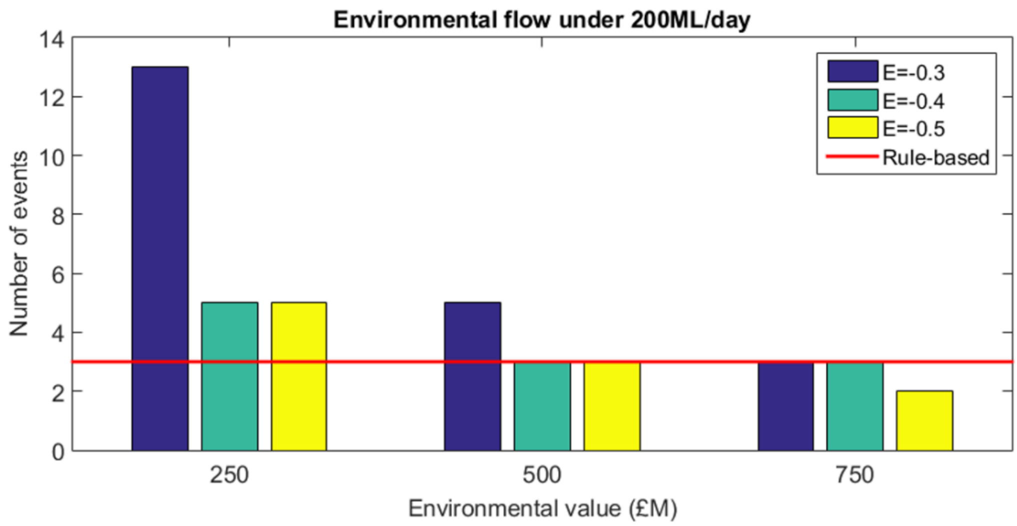


Figure 13. Duration and number of occurrences of weeks with environmental flows below 200 ML/day, scenario 1.

Impact on prices varies depending on the case (Figure 13 to Figure 15). Prices increase less when the price response is better (high elasticity) but increase more when prices have to match a higher value of water in the river. Prices increases of at least 10% are frequent, sign that scarcity is ubiquitous in the 2050 Thames River basin. At the other end of the spectrum, price increases of 150% happen roughly as often as no-flow situations: they are only warranted in extreme situations. One can imagine scarcity pricing to be then complemented by regulatory restrictions to avoid ecological catastrophes. This means that a 150% price increase would not have to happen in practice. In fact, 100% price increases would only happen in cases with low elasticity and medium or high environmental value, or with medium elasticity and high environmental value. In the other three cases, 100% price increases would rather correspond to the severe scarcity situations: they could be used. As for 50% price increases, they correspond to severe scarcity situations in the six other cases.

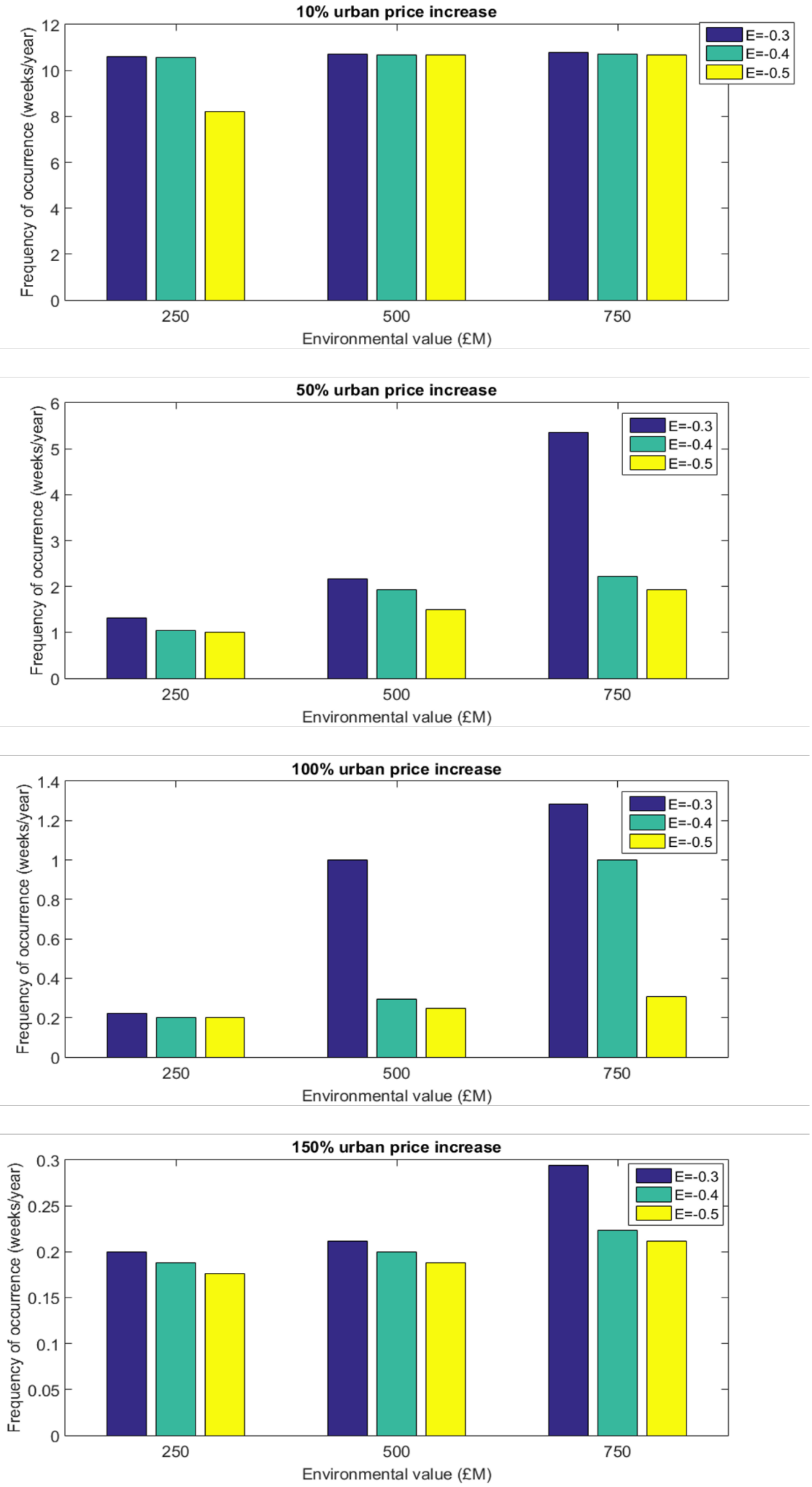


Figure 14. Frequency of price increases, scenario 1.

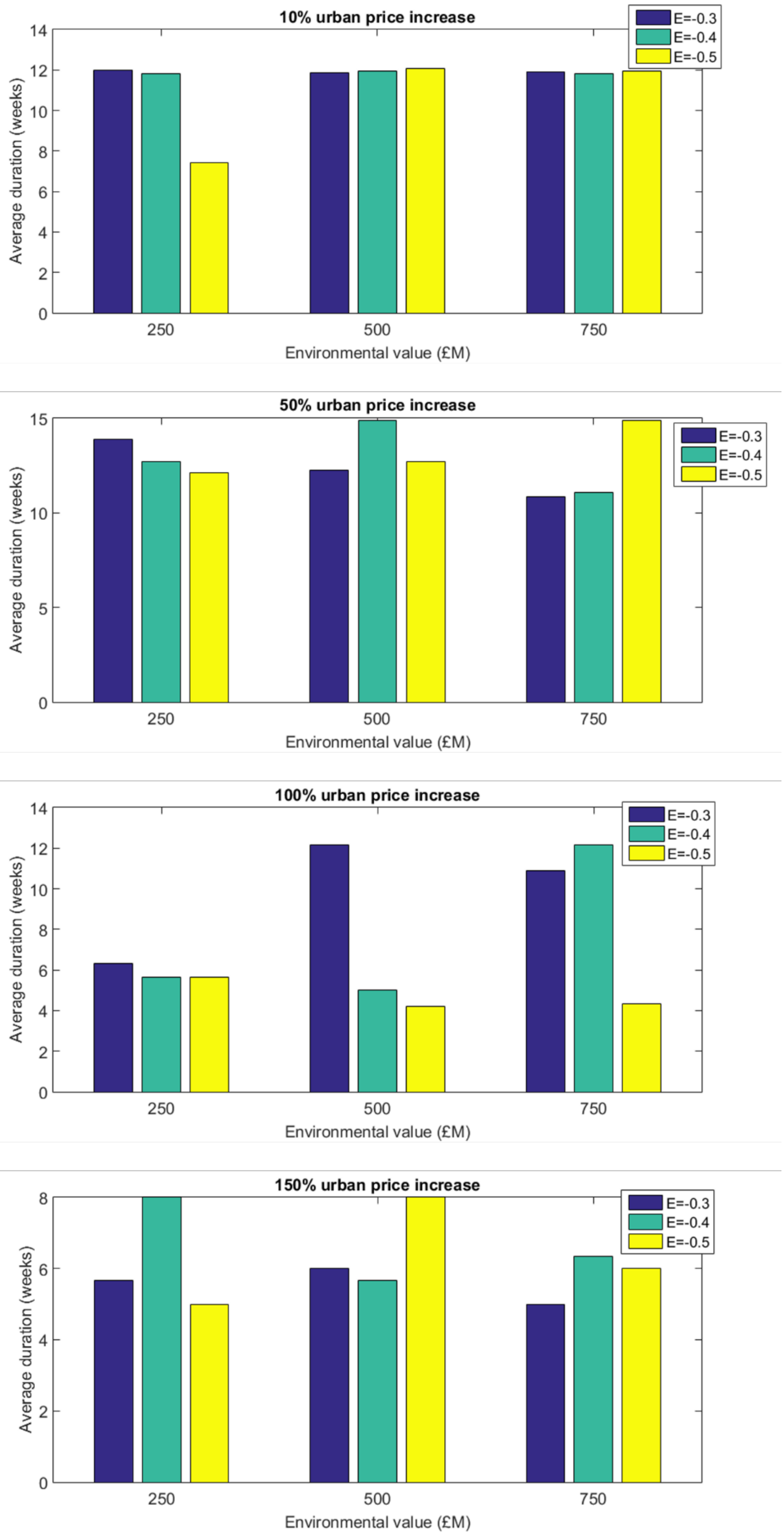


Figure 15. Duration of price increase events, scenario 1.

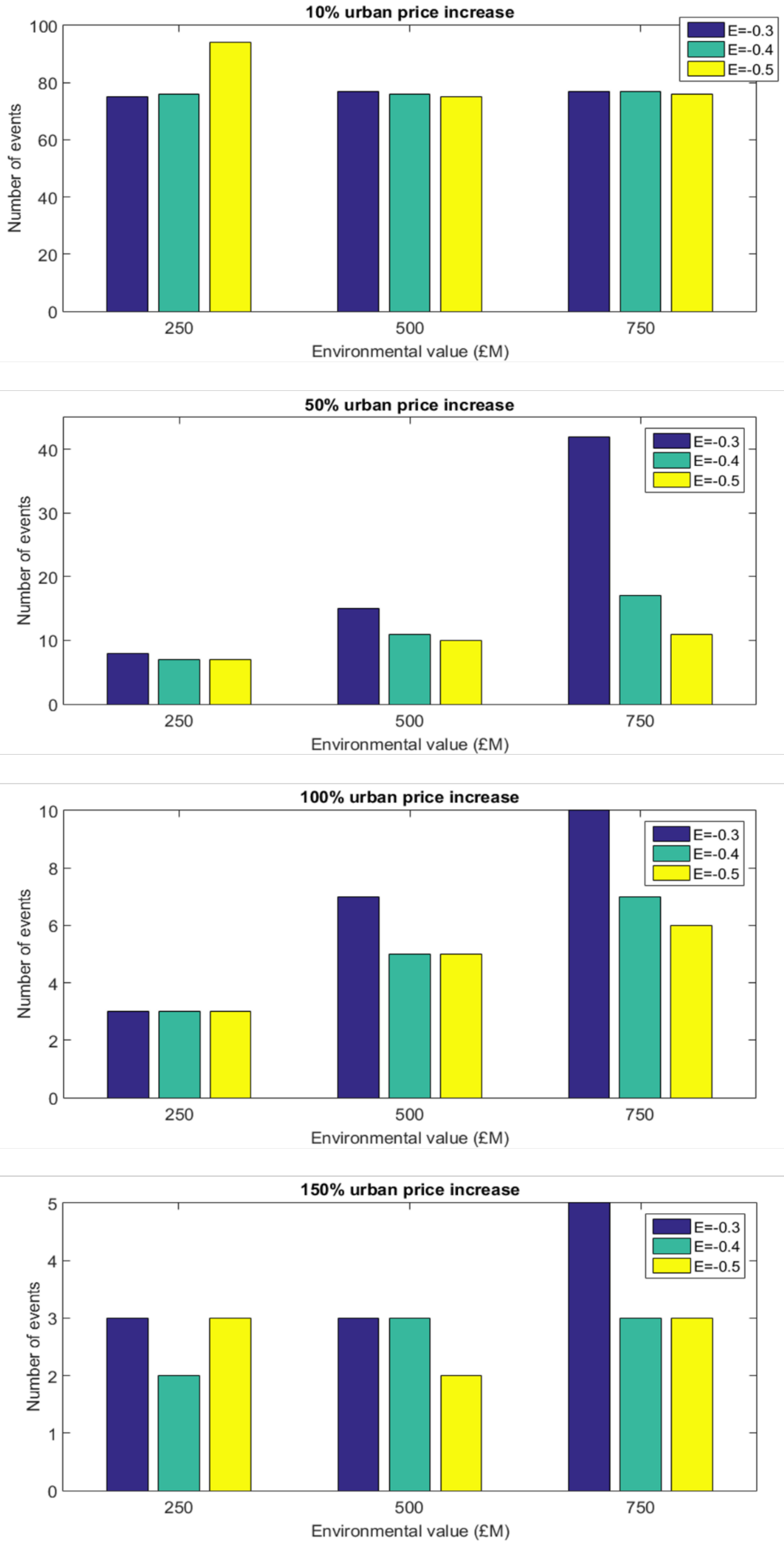


Figure 16. Number of price increase events in 85-year period, scenario 1.

4.3.2 *Scenario 2: residential demand curve with constant demand slope*

Contrary to scenario 1, in scenario 2, it does not get more difficult to achieve a reduction in consumption when residential prices are already high. As a consequence, it is expected that the efficient price that yields the same MROC for the two water uses – urban and environmental – will be lower given the same amount to allocate. This mechanically lead to lower environmental deficits (Figure 17). Deficits are lower by 15 ML/day with £250M environmental value regardless of elasticity, and by 21 ML/day with £500M environmental value, still regardless of elasticity. Elasticity does have an impact in the high-value case, where the difference goes from 22ML/day when E=-0.5 to 27 ML/day when E=-0.3. This also means that the relative decreases become higher when the environmental value becomes higher. In contrast, it is at its mean when the environmental value is low.

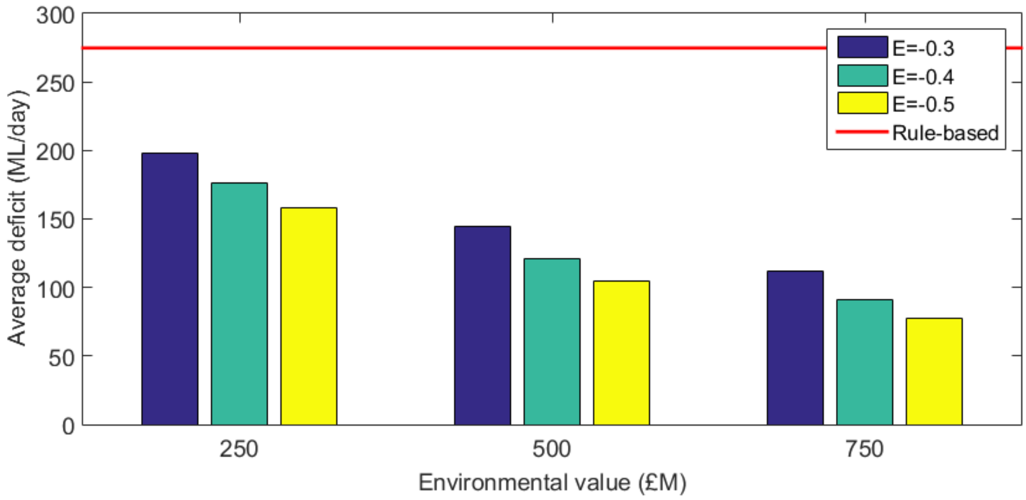


Figure 17. Average deficit for weeks with an environmental flow deficit in the rule-based simulation, scenario 2.

Likewise, the impact of the shape of the demand curve on the occurrence of severe scarcity events (below 400 ML/day during a week) is minimal in the low-value cases (Figure 18). For medium and high environmental values, the impact of increased environmental value on reducing both the occurrence and duration of events is greater than in scenario 1.

When it comes to extreme scarcity events (under 200ML/day during a week), the impact of both environmental value and water scarcity becomes clearer in scenario 2 (Figure 19). Six of the cases yield result that are quantitatively a little better than for scenario 1. Yet for the three remaining cases, the best combinations of high environmental value and high elasticity, extreme scarcity events are eliminated (in two cases) or reduced to one occurrence of only two weeks – better than in rule-based simulations. On the other end of the spectrum, in the three low-value cases, the number of no-flow weeks is nearly as bad as in scenario 1, as it ranges from 14 to 17. In the medium value cases, it falls to 2 weeks altogether for the low-elasticity case, then no-flow occurrences all but vanish for the 5 remaining cases.

Overall, these results indicate that the environment is more protected when the demand reacts linearly to price changes. In a way, this suggests that having the same shape for the environmental demand curve as for the residential one could help design scarcity pricing schemes that better protect environmental flows when they have enough value compared with the collective willingness to pay of the users from the urban area they are competing for water with, and when the prices response of this urban water user is good enough. Yet, a linear demand curve is a behaviour that seems unrealistic for residential users, when restrictions are important. Its results are consistent with the more sophisticated demand curve of scenario 1 for smaller price increases, but its validity may not extend beyond.

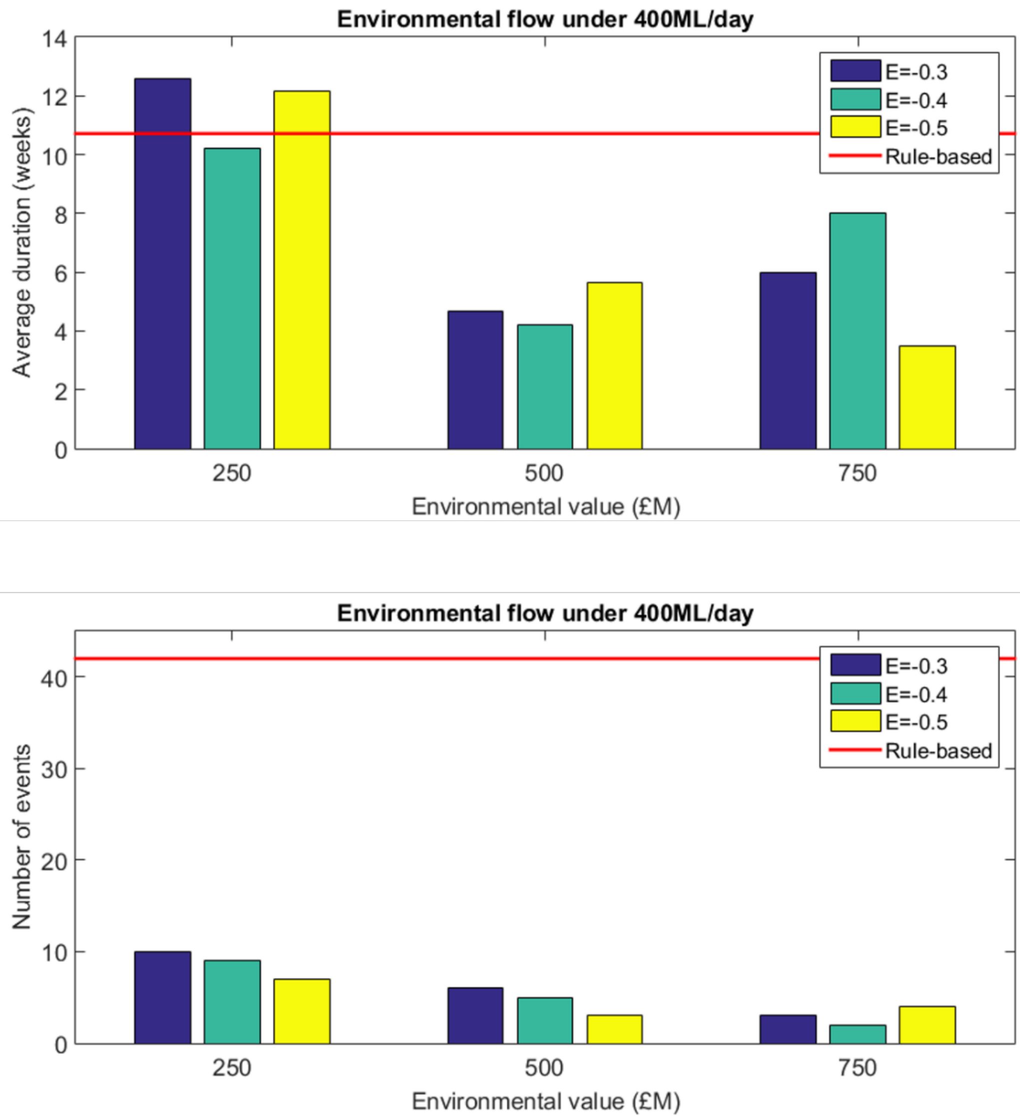
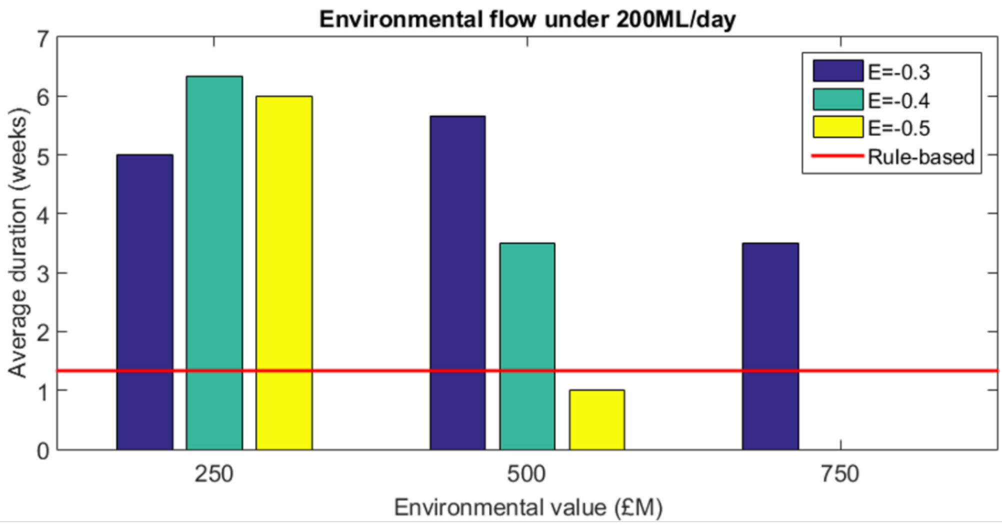


Figure 18. Duration and number of occurrences of weeks with environmental flows below 400 ML/day, scenario 2.



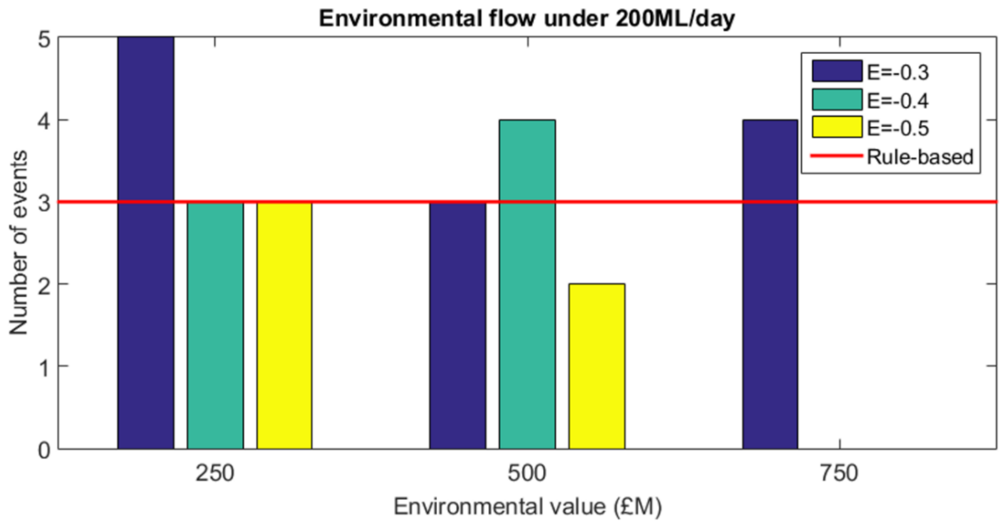


Figure 19. Duration and number of occurrences of weeks with environmental flows below 200 ML/day, scenario 2.

When it comes to the impact of scarcity pricing on water prices (Figure 20 to Figure 22), similar remarks as for scenario 1 hold, even though price increases tend to be a little less pronounced because of the shape of the residential demand curve. In particular, similar to scenario 1, results indicate an upper bound on the potential price increases. For instance, 150% price increases cause either so much demand reduction that they are not warranted, or are very rare. Again, the realism of the price response for such big increases is questionable with a linear residential demand curve. Such price increases would lead to demand reductions of 45%, 60% or even 75% depending on the value of E.

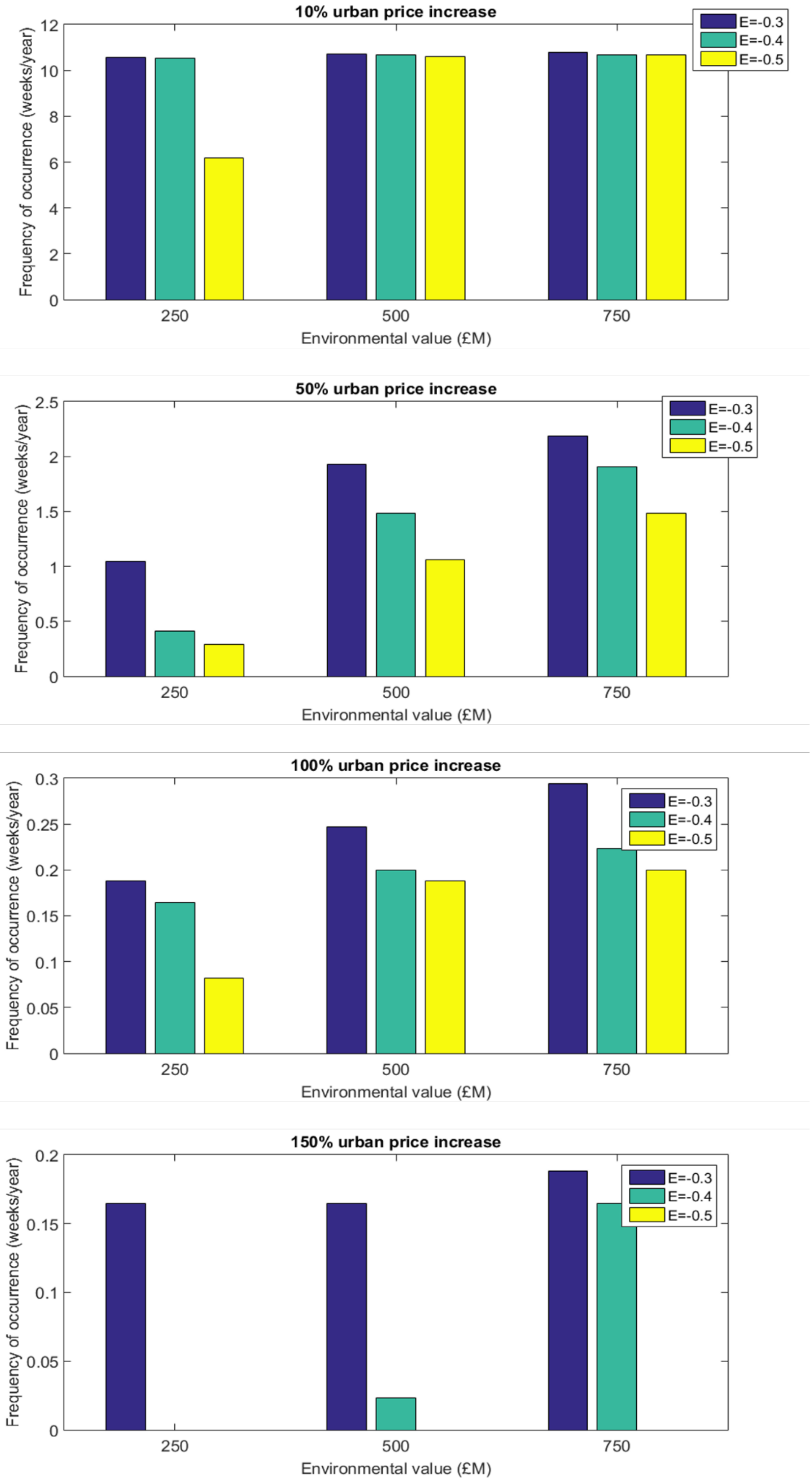


Figure 20. Frequency of price increase, scenario 2.

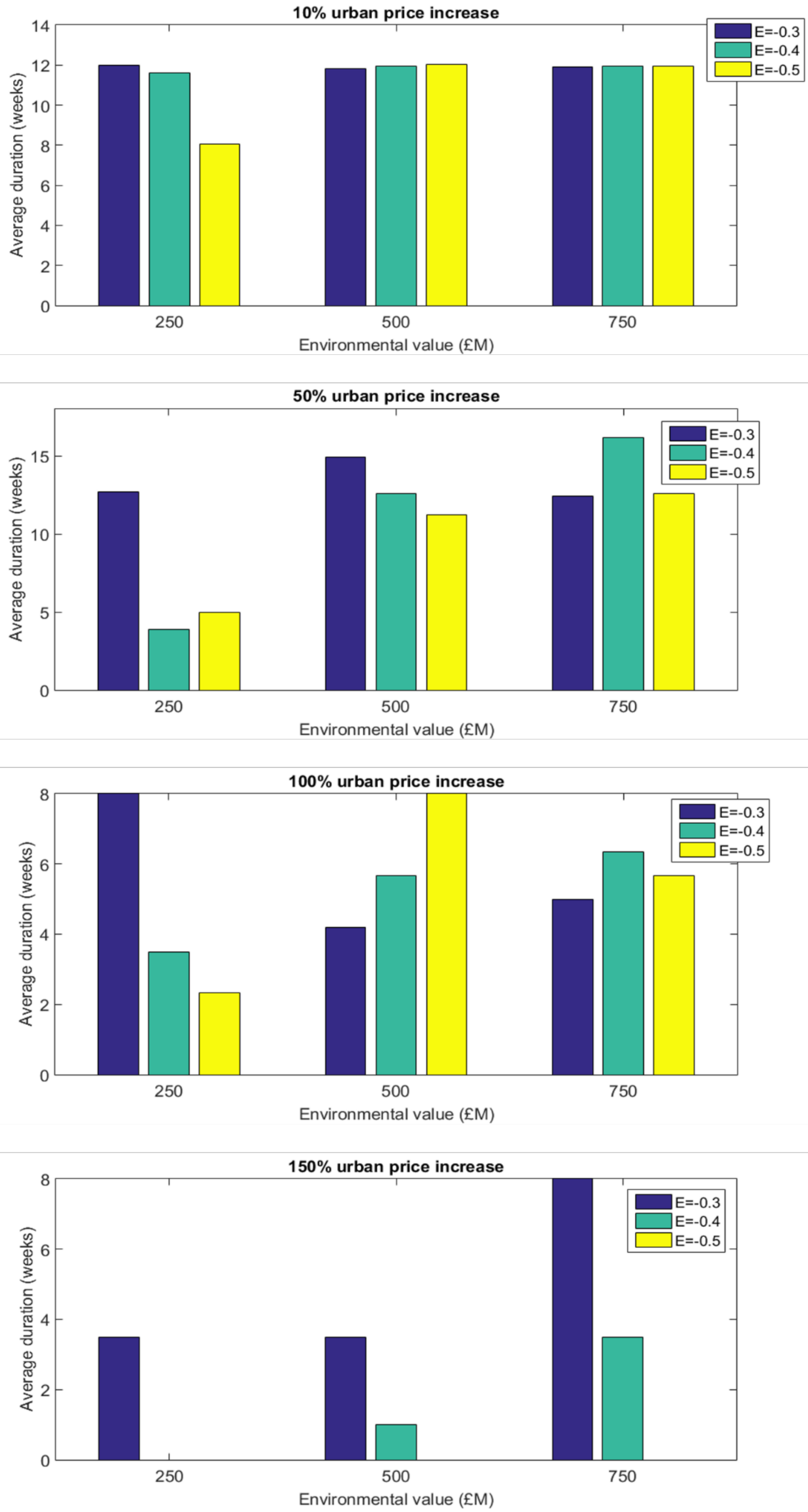


Figure 21. Duration of price increase events, scenario 2.

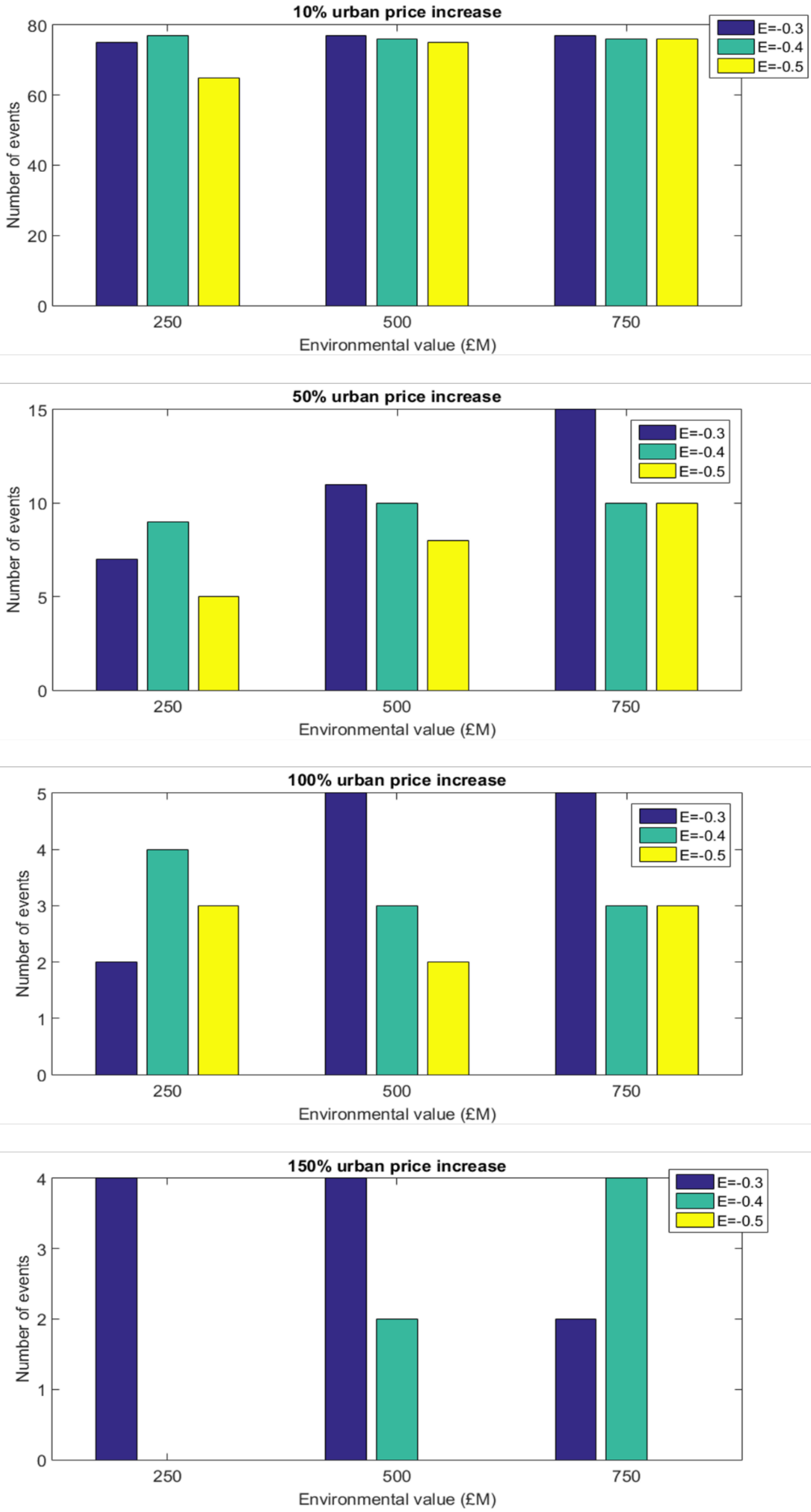


Figure 22. Number of price increase events in 85-year period, scenario 2.

4.3.3 *Scenario 3: including desalination costs (constant elasticity case)*

This scenario investigates the impact of adapting the treatment and delivery costs to the cost of treating and delivering water using the most expensive technology under use. This is another way to give a signal that water is scarce and should be used less. In this scenario, we assume that the fact of using the desalination plant results in a marginal cost of treating and delivering water that is £0.30 per m^3 more expensive than the use of other means. This value of £0.30 per m^3 is reasonable in a context where the price of desalinated water varies widely with the price of energy [Zhang and Babovic, 2012]. It is not meant to be accurate, in the context of a forward-looking study of the future London water supply. Rather, it is meant to investigate how dynamic pricing could be used to reflect the different costs incurred by the water utility when water becomes scarce and difficult to access.

In scenario 3, the most expensive water source becomes the one used to set the marginal price of residential water. River water becomes more expensive than desalination when water becomes scarce enough that its value exceeds the additional cost due to desalination (Figure 23). Recall that desalination is triggered when Thames River flow at Teddington weir drops below 3000 ML/day, which is much above the minimum environmental requirement of 800 ML/day.

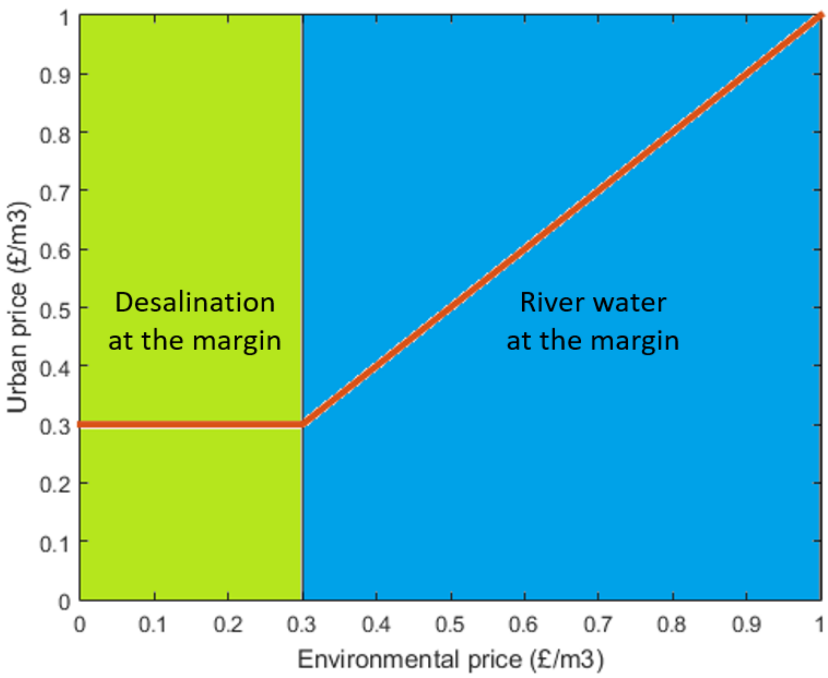


Figure 23. Water use at the margin depending on environmental price.

Therefore, the impact of this type of scarcity pricing on environmental flow preservation is minimal, as it only has an impact on consumption when environmental flow value is below £0.30 per cubic meter. Across the nine cases, the maximal average deficit reduction compared with scenario 1 was under 5 ML/day. Besides, it has no impact on severe or extreme scarcity events. The impact of this type of policy would be better appreciated in places where large reservoir storage would enable to store the water saved when desalination is at the margin.

5. Valencia case-study

This section presents the supply-demand modelling and its results on scarcity pricing from the Valencia case-study. The Valencia context is presented in Section 5.1, showing frequent water scarcity in the area where agriculture is the main water use. Case-study objectives are also detailed in that Section. Then, Section 5.2 presents the hydro-economic modelling leading to the evaluation of scarcity pricing in that context. Finally, Section 5.3 presents the results of this modelling exercise.

5.1 Context and objectives

5.1.1 Presentation and context

The case study is the supply to the city of Valencia from the Jucar River system, a complex water resources system located on Eastern Spain. This is one of the 9 exploitation systems of the Jucar River Basin District, which includes Cenia-Maestrazgo, Mijares-Plana de Castellón, Palencia-Los Valles, Turia, Júcar, Serpis, Marina Alta and Marina Baja-Alacantí Vinalopó (Figure 24).

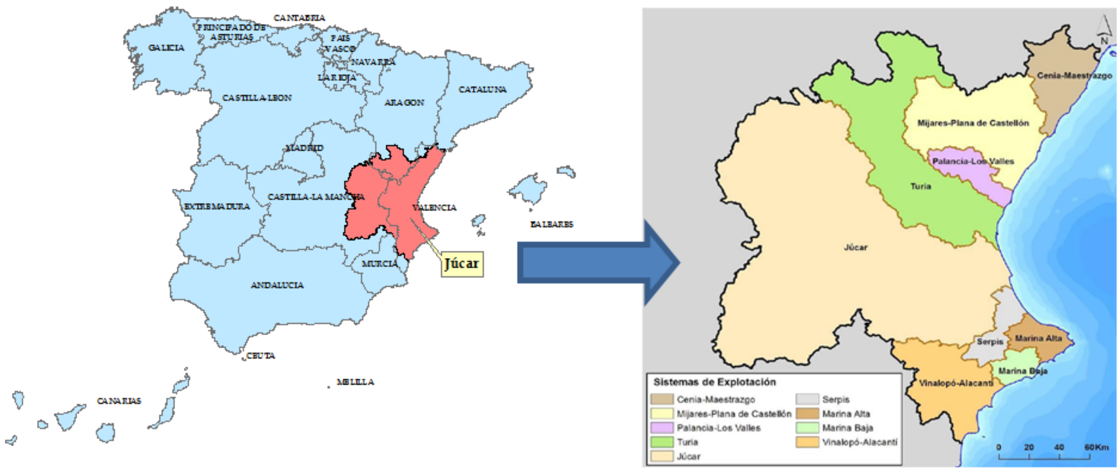


Figure 24. Situation of the Jucar river basin.

The Jucar system is the largest one (2,2378 km^2) and comprises the Jucar River basin and the area supplied by the Jucar- Turia canal and coastal sub-basins ranging from Gola El Saler to the limit of Cullera and Tavernes de Valldigna villages. The hydrographic network consists of the Jucar River and its tributaries (Cabriel, Valdemembra, Arquillo, Magro, Albaida, Sellent, etc). The basin is highly regulated and with a high share of water for crop irrigation (about 83%; Figure 25). Total water demand in the Jucar River basin is about 1650 Mm^3/year, whilst the average water resources availability is 1747 Mm^3/year (from 1940/41 to 2008/09) (CHJ, 2013). This means that demand exceeds supply on a dry year, and that interannual reservoir storage and groundwater use are the only buffers for drought then. The urban demand corresponds to the 13 % of the total water demand.

The municipality of Valencia receives water from two water treatment plants, “La Presa” and “El Realón” (45 % and 55 % of the total demand respectively) (Figure 26) and from 2 rivers, Jucar (through the Jucar-Turia canal for interbasin transfer; 75% of the supply on average) and Turia (25%).

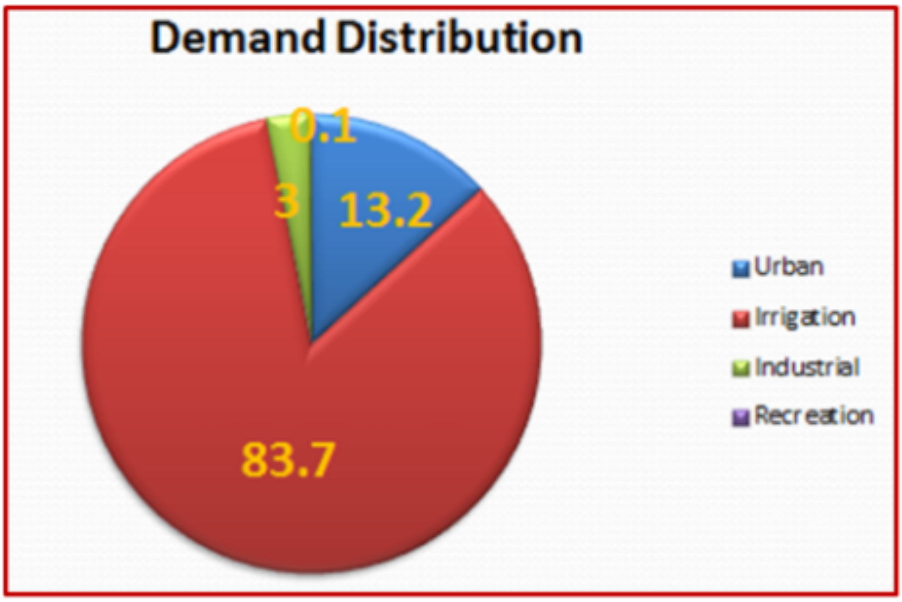


Figure 25. Demand Distribution in the Jucar river bain.

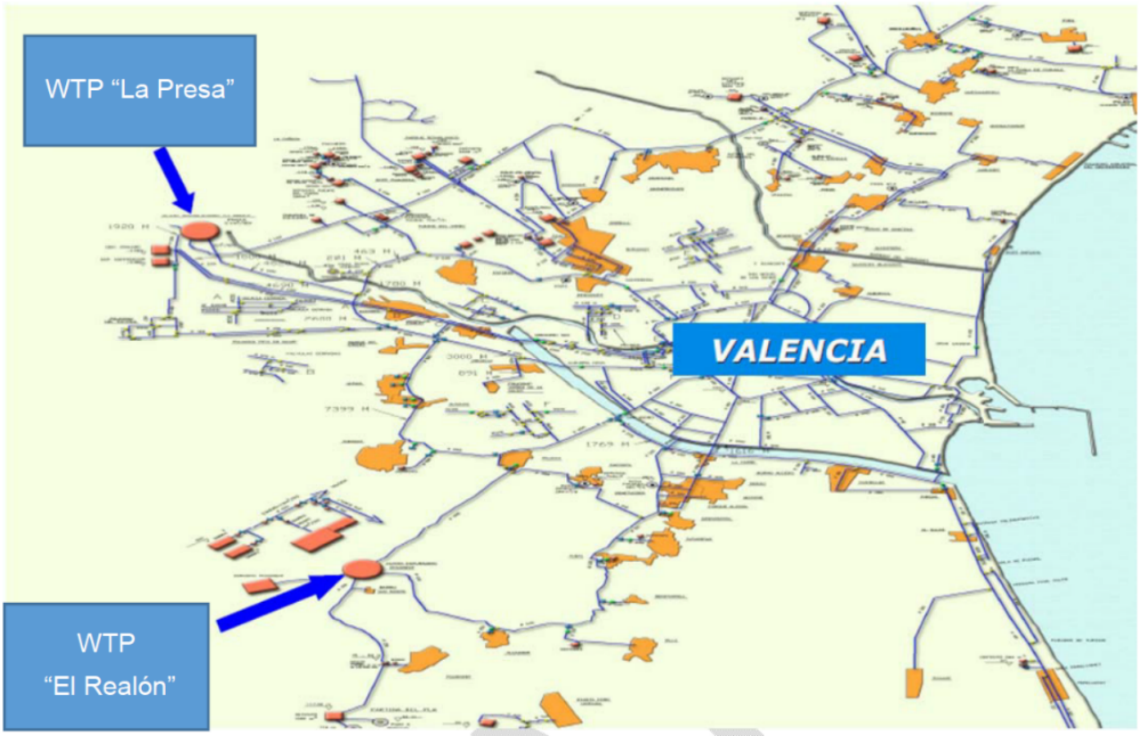


Figure 26. WTP for Valencia city (Source: adapted from CHJ, 2010)

5.1.2 Objectives

The objective is to design water rate structures for urban demand combining increase block rates and dynamic pricing, including a water scarcity component derived from the marginal value of water at river basin scale. This marginal value is obtained through hydro-economic modelling of the river basin system, including the main supply sources and the demands, economically characterized through demand functions. The use of a component of the pricing scheme corresponding to the marginal value of water allow to send the users a signal of the economic value of the resource (related to water scarcity) and the opportunity cost of water use, promoting a more efficient allocation and use of water. In this way, pricing will be design under a double goal: as a cost recovery tool (financial instrument), but also as an economic instrument for demand management.

The proposed water rate performs the expected basic functions for water tariffs [Hanemann, 1998; Griffin, 2006]:

- **Generate revenues**: the proposed rate produces sufficient revenues to allow the utility to cover its costs and meet its financial requirements, in both short-run and long-run conditions (revenue sufficiency and neutrality).
- **Allocate costs**: the total costs are allocated among the users under a certain rate structure. The water rate should be perceived as affordable, fair and equitable by the users.
- **Provide incentives** sending a signal to the users in scarcity situations that performance as incentives to use water efficiently.
- **Simplicity and legality**: the proposed rates should be easily understood by the clients and utilities and legally acceptable.

5.2 Methodology: hydro-economic modelling

5.2.1 Modelling framework for getting the marginal value of water at source

According to the economic theory, efficient pricing should integrate scarcity values, sending the users a signal of water scarcity and the economic value of water [Griffin, 2006].

In the same way as in Section 2.2.1, MROC is here defined as the value of an additional unit of water in the system at the source. Since current water prices do not respond to market conditions, water values need to be estimated through hydro-economic models (HEMs) and certain assumptions on the economic value of water for each use (demand curves). HEMs allow an integrated analysis of water supply, demand and infrastructure management at the basin scale [Pulido-Velazquez et al. 2008, Harou et al., 2009]. One of the advantages of HEMs is the explicit treatment of the economic value of the water, generally represented using mathematical functions that relate the marginal economic cost of the water with the quantity of water consumed, or water demand function (see Section 3.1.2). The MROC or marginal value of water at different reservoirs in the system can be determined through the shadow values obtained by the economic optimization of water allocation in the system, or using a simulation approach that takes into account the priorities and the system operating rules in reservoir releases and water allocation decisions.

Figure 27. General flowchart of SIMGAMS tool shows the general flowchart of the method proposed to get the marginal value of water at source at river basin scale. First, the river basin is characterized in terms of topology of the flow network, available infrastructure, hydrology and economic information. For that purpose, we have used HydroPlatform [Harou et al., 2010] as an auxiliary tool; this is an open-source software platform for network (node-link) model that allows input, storage, display and export model data, including the connectivity matrix of the system (which represents the relations between nodes and links).

Water management tools are required to assess water scarcity costs at river basin scale; in this case we used a new DSS tool SIMGAMS [Lopez-Nicolas, 2014]. SIMGAMS uses a simulation approach, allocating water resources on a monthly basis according to different water management targets (on demands, storage, environmental flows, etc.), priorities and other system operating rules (such as reservoir operating rules). The main results are water management (storages, deliveries, etc.) and economic results (like water scarcity cost and marginal resource opportunity costs (MROC).

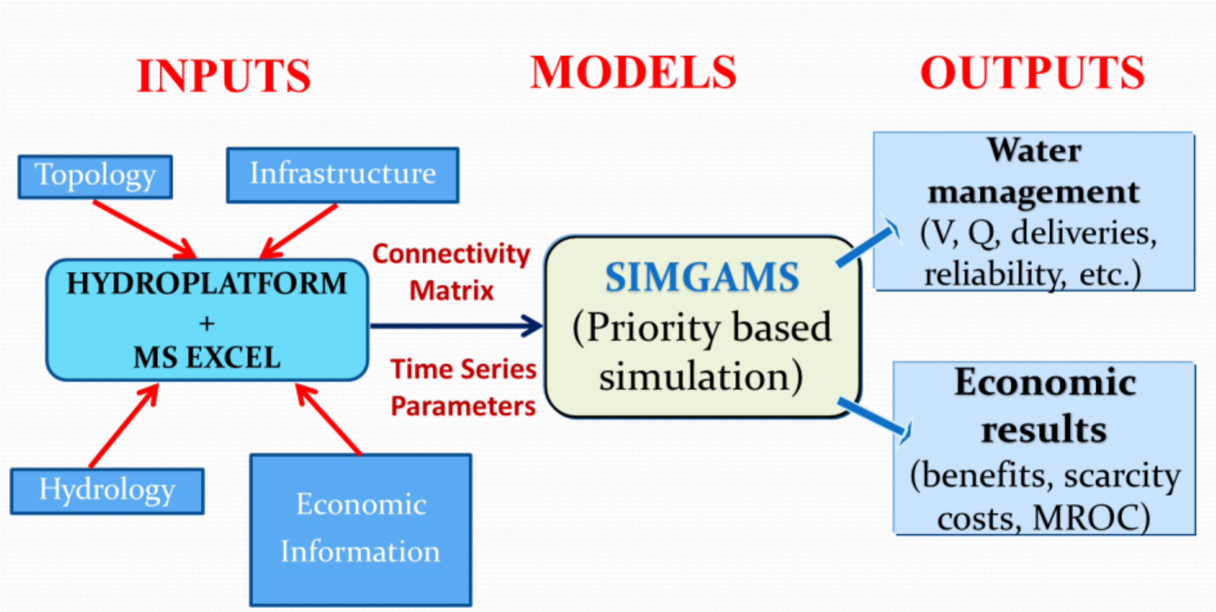


Figure 27. General flowchart of SIMGAMS tool.

After allocating the available water resources among the competitive users, SIMGAMS evaluates the associated water scarcity cost for each user of the system through the economic functions of each user of the river basin. Scarcity costs can be interpreted as the area under the demand curve, for the range of water deliveries greater than the actual demand (Figure 28), when the cost is that of the commodity itself: this corresponded to the case of the "raw" water demand curve on Figure 3. In other words, scarcity costs is the area under the curve that is not the consumer's gross benefit (see Section 3.1.2), when no transaction or other costs are considered.

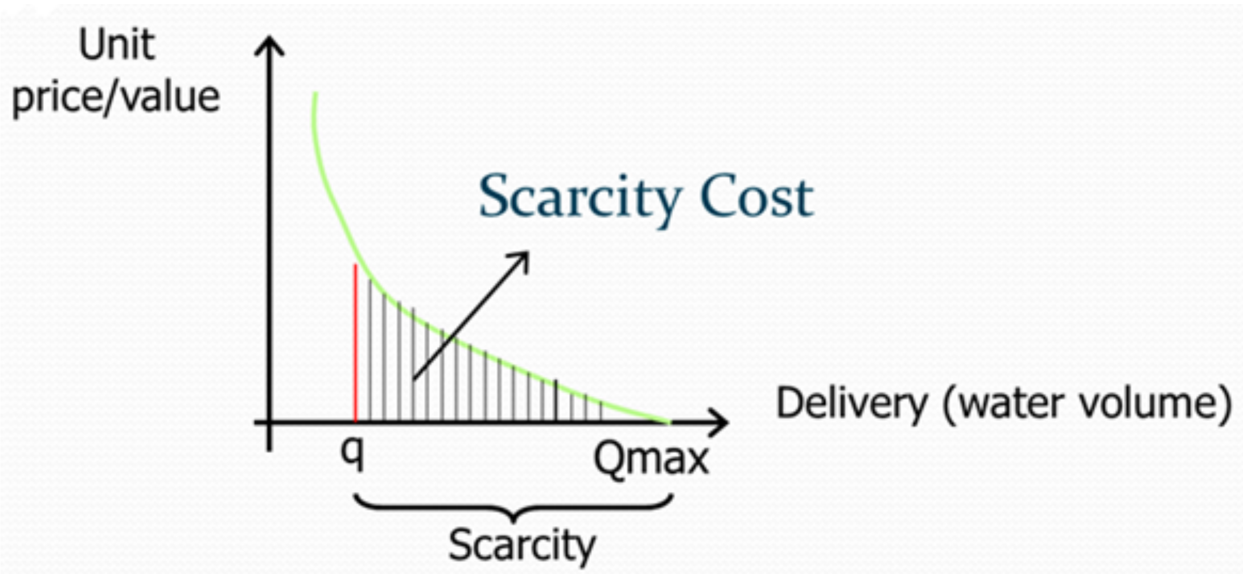


Figure 28. Scarcity costs in a demand function.

The aggregated water scarcity cost time series at river basin scale constitutes the basis for getting the MROC values:

- First, calculation of the baseline benefit or water scarcity cost of water use at the river basin scale, according to priorities and system operating rules (simulating approach);
- Secondly, evaluation of the new benefit or cost, at each time, after applying an extra unit of water resources at a specific location (e.g. reservoirs) (at source);
- Finally, assessing the extra benefits due to the more efficient allocation of water resources, approximating the MROC by the quotient between the benefit difference and the difference in water allocation:

$$MROC_t = \frac{Benefit_t - Benefit_{baseline}}{\nabla Volume}$$

5.2.2 Model of the Jucar River basin

River basin

The network of the Jucar river basin has been developed using as main sources the data from the Jucar river basin agency, e.g. [CHJ, 2013] and previous studies and models of the basin developed with SIMGES, AQUATOOL [Andreu et al. 1996]. Figure 29 and Figure 30 show the Jucar river basin network that has been developed using HydroPlatrform.

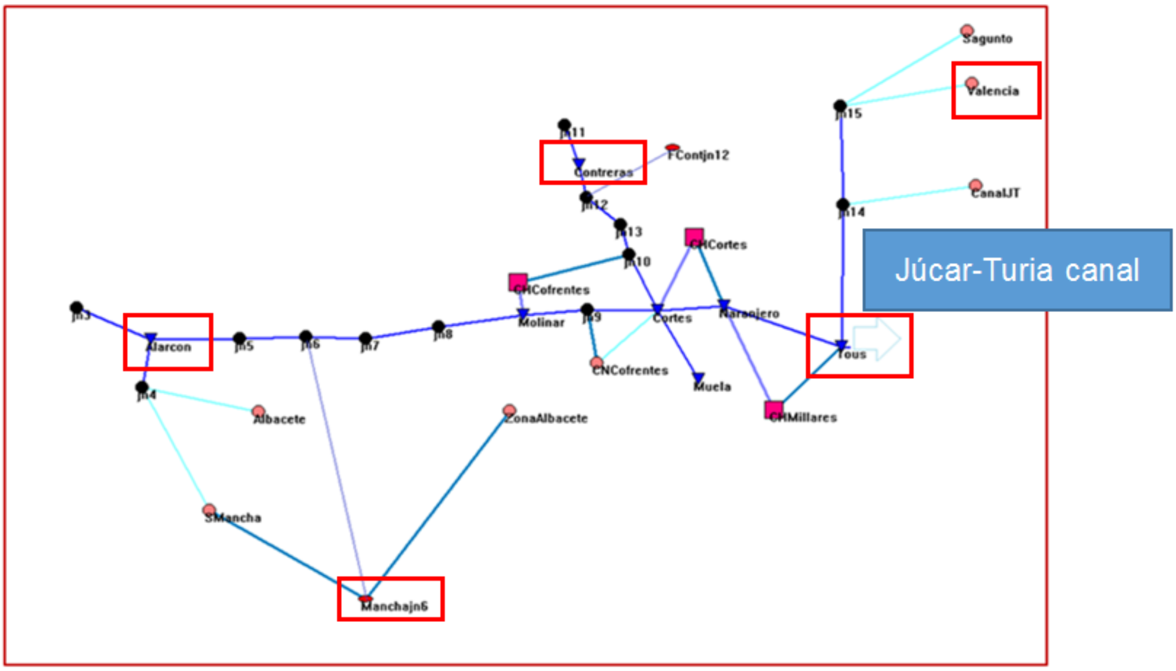


Figure 29. Jucar river basin (1/2)

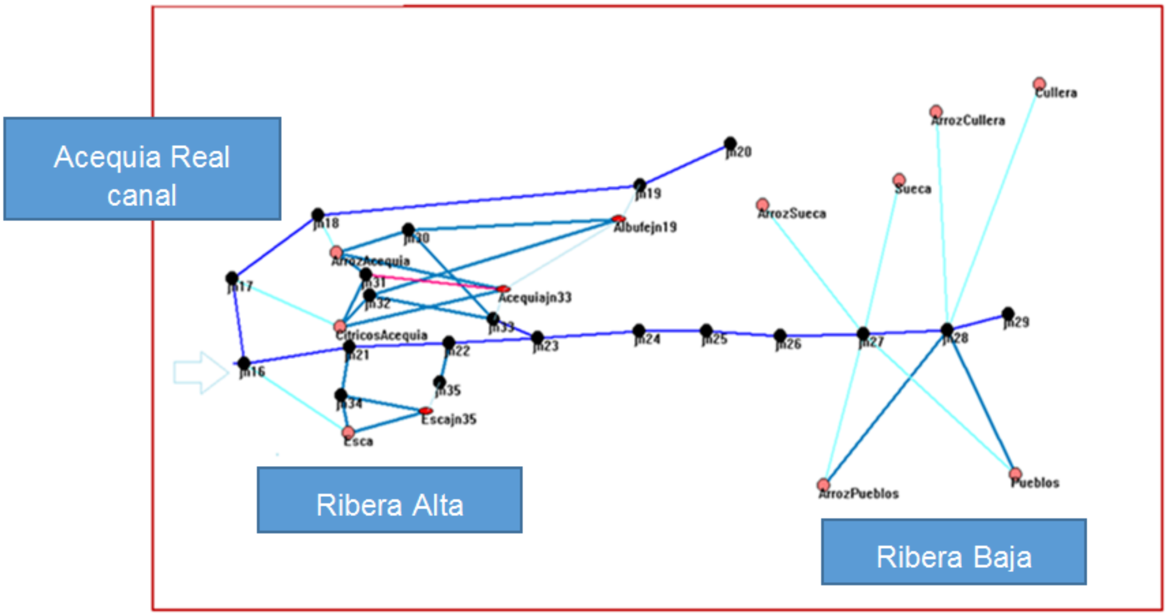


Figure 30. Jucar river basin (2/2)

The main reservoirs in the Jucar river basin are Alarcón, Contreras in the upper basin, and Tous in the mid basin. Total storage is worth over a year of average annual flow (Table 5; for Contreras, actual storage capacity is 852 Mm^3, but useful capacity is much lower). The Jucar river basin agency develops the system operating rules considering Alarcón and Contreras as the main overyear storages. Tous reservoir is used mainly for flood control and for seasonal intrayear regulation during the irrigation season.

Reservoir	Storage capacity (Mm^3)
Alarcón	1,112
Contreras	440
Tous	378
Others	231
Total	**2,261**

Table 5. Storage in the Jucar River basin.

The most important irrigation infrastructure are the Jucar-Turia and Acequia Real canals. The Jucar-Turia canal supplies water to the urban demands of Valencia (the municipality and its metropolitan area) and Sagunto; and the Jucar-Turia irrigation district. The Acequia-Real canal supplies water to the Ribera Alta district (mainly for rice crops and orange trees). The Ribera Baja irrigation district is located in the lower basin; the main crops are rice and orange trees, as in the Ribera Alta.

Municipality of Valencia

The city of Valencia has around 800,000 inhabitants, and it is fully equipped with smart water meters at almost each households (around 430000 clients approximatively),. Figure 11 shows the distribution of households per block tariff.

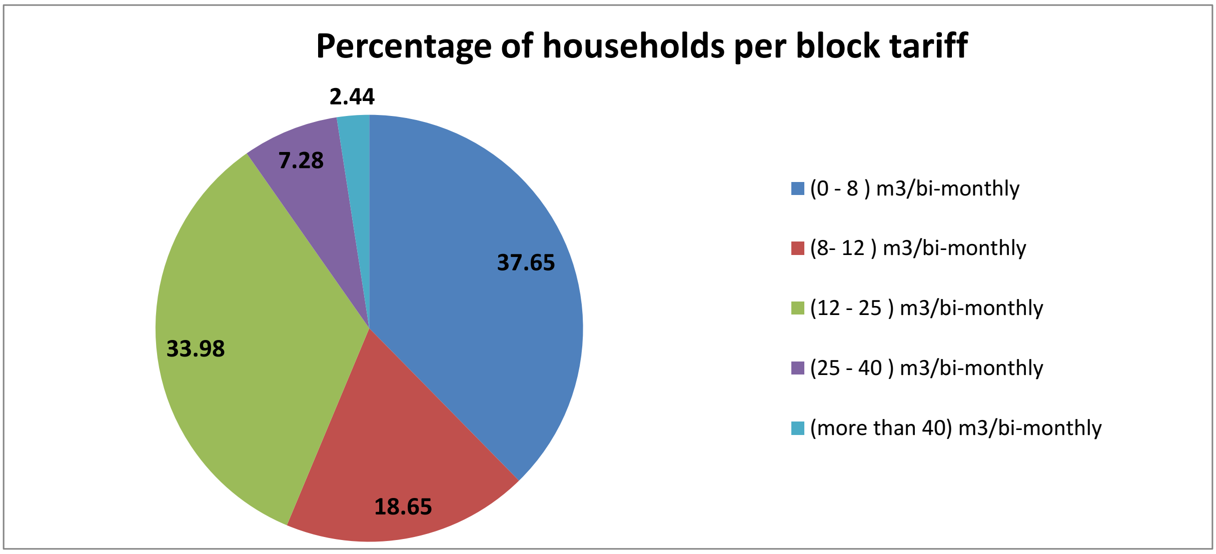


Figure 31. Households per block tariff

The price-elasticity for Valencia is estimated as -0.64, coming from a study led by Garcia-Valiñas [Garcia-Valiñas, 2004; Garcia-Valiñas 2006] for the Spanish Ministry of Environment and Agriculture, provided an econometric estimate of the price-elasticity of water demand for urban supply in the Valencian region within the Jucar river basin district. The urban water demand, excluding urban municipal and agricultural uses, was estimated using a log-linear functional form. With the average annual per capita volume of water (m^3) as independent variable, the model incorporates the following explanatory variables: price (average cost), income per capita, percentage of single-family homes, percentage of vacation homes, a binary variable to identify coastal homes, and two variables for identifying industrial and tourism activities.

5.2.3 Design of dynamic scarcity-based urban water tariffs

The proposed approach follows a two stage process; first, we design a water tariff without including MROC values (corresponding to a non-scarcity situation), baseline water tariff, fulfilling the expected basic functions for water tariffs, e.g., [Molinos-Senante, 2014]. Then, this water tariff is used as a basis to develop the water rates that take into account the MROC associated with scarcity situations. We propose two approaches for this second stage: the first one takes into account the MROC time series; the second one adopt a practical approach, based on predefined targets of water savings during scarcity in order to meet the general aims of the River Basin Management Plans.

Stage 1

The design of this baseline water rate structure requires the assessment of the real water supply costs. This requires:

- Assessment of the fixed and variable cost associated to water treatment and distribution (in M. € / year and €/m^3 respectively) (at user level)

- Design of the pricing steps equating the annual income to the total cost, considering for that the volumetric charge (€/m3) and the fixed rate (mainly service and maintenance charges) to the total cost. For that, the starting point is the current official water rate structure, considering the number of blocks and distribution of households per block tariff. For example, Figure 32 shows the current water rate structure for the Valencia case study (for most households, except from large families). It follows an increasing block rate structure, with a first block at a discounted rate to enhance the efficiency of water consumption and promote water conservation (the large users are penalized). On the other hand, it has an equity aim, in the sense that large users (often wealthier) are subsidizing the basic uses for everybody.

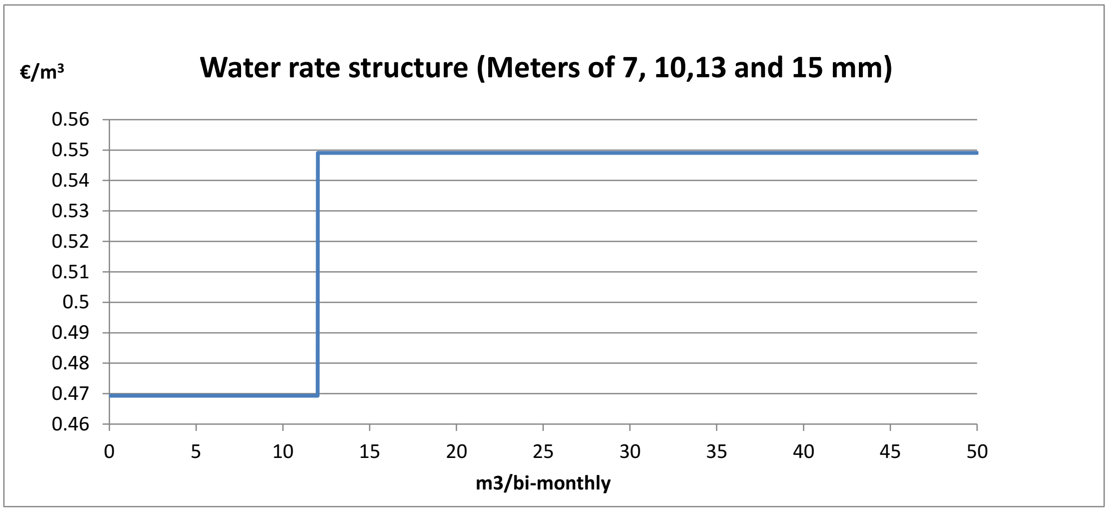


Figure 32. Valencia water tariff (Source: data from Diari Oficial CV, 2015).

The threshold of the first block, 12 m^3 in 2 months, has been defined based on equity reasons. According to the World Health Organization [Guy et al. 2003], a basic requirement of per capita water use is 100 l/day/person. Moreover, 60 % of the households have 2 or less persons in Valencia case study. So, the consumption for households of 2 persons corresponds to 12 m^3 per 2 months.

The revenue sufficiency or revenue neutrality condition states that income I must equal the sum of costs:

$$I = \sum_{s=1}^{n} C_s$$

Where the income can be decomposed into the revenues from each block, or each individual group of users:

$$I = \lambda_1 \cdot \sum_{i=1}^{n} V_i + \lambda_2 \cdot \sum_{j=1}^{m} V_j + FI$$

where λ_1 is the price for the first block, λ_2 is the price for the second block and FI the fixed income. Likewise, costs can be decomposed as follows:

$$C_s = FC + \sum \frac{V_i}{E} \cdot VC$$

where "s" is each "water source", FC are the fixed costs, E the efficiency and VC the variable costs.

Then, using the revenue neutrality condition, it is possible to get the values of the second block price λ_{2_b} if the values of the first block price λ_{1_b}, derived by the utility from expert knowledge, is known.

Stage 2

The values of $\lambda_{1\mathrm{b}}$ and $\lambda_{2\mathrm{b}}$ from stage 1 are the basis to develop the new water rate structures in the two following approaches. These water tariffs will be dynamic depending on the storage at the reservoirs, but fixed for each year. In other words, they can change several times in a few months if storage happens to be very variable in a given year, and that requires smart metering to enforce; yet, users can get to know beforehand what prices may be enforced, and react accordingly. The criterion is to adopt a dynamic tariff depending on the available storage in October 1st (beginning of the hydrological year). We consider that "λ_{1b}" (for the first block) and the consumption threshold (12 m^3 bimonthly) remains constant due to the equity condition.

Approach1. Adding MROC to the baseline prices during scarcity periods

The baseline pricing guarantees revenue sufficiency (financial cost recovery). The second design criterion is to cover the opportunity cost during that scarcity situations by adding the MROC (marginal water value at the source) to the baseline price for the 2^{nd} step, according to the available reservoir storage:

$$\lambda_{2i}^{'} = \lambda_{2b} + MROC_i$$

where $\lambda_{2i}^{'}$ is the new volumetric charge associated to each MROC step and *"i"* each step of the MROC pricing function.

Water demand will change as a reaction to the different values of the second block price $\lambda_{2i}^{'}$ (with respect to the associated demand to baseline"λ_{2b}" value from step 1). The modification of the demand will be given by the price-elasticity of demand, as defined in Section 3.1.1. Then, that modification in the demand will in turn affect income and costs. Therefore, the price elasticity enables simultaneously assessment of the shifts in income and costs.

Approach 2. Practical approach considering predefined water saving targets

This second approach is based on predefined water saving targets to reduce loses during scarcity conditions, fulfilling also the sufficient revenue criterion. We have defined four scarcity levels, linked to the reservoir storage ranges (Table 6).

Volume ranges	Scarcity situation	% Water saving targets
0- V1	Worst	A %
V1 to v2	Intermediate	B %
V2 to V3	Regular	C %
> V3	Good	0 %

Table 6. Scarcity scenarios.

Then for each scenario we find the minimum second block price that meets the water saving target while meeting the revenue sufficiency condition. Simultaneously, we derive the new revenue and costs from the reduced demand, similarly to what is done in Approach 1.

Figure 33 shows an example of the dynamic water rates designed with this practical approach, showing the different volumetric charges associated to different percentages of water saving targets.

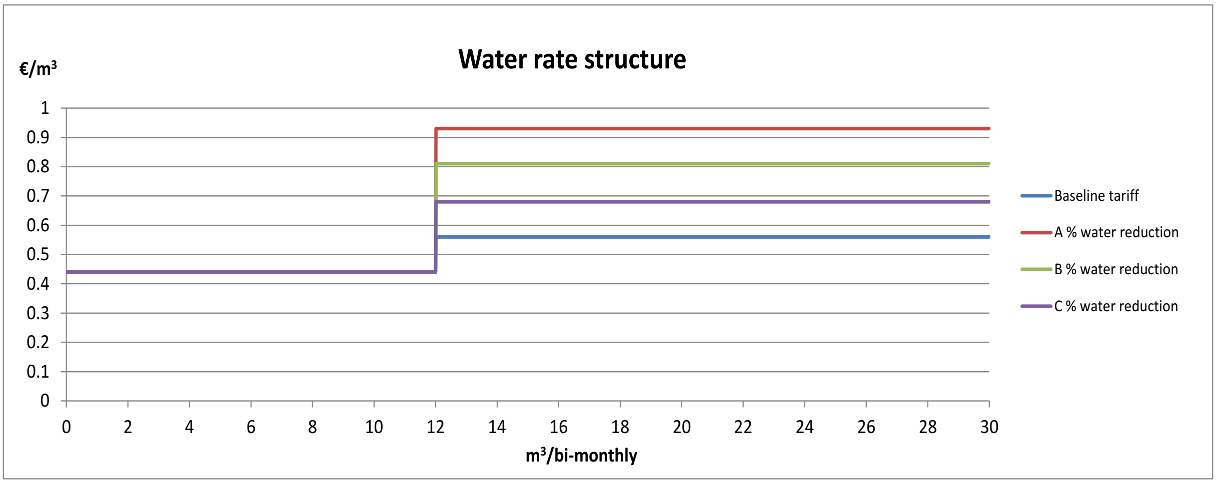


Figure 33. Urban water rate structure

5.3 Results

This section illustrates the main results of the Valencia case study.

5.3.1 Simulating MROC

The MROC simulation approach has been applied to get the marginal value of water in the Jucar river basin, considering the main reservoirs of the system. Figure 34 and Figure 35 show the results for Alarcón reservoir that is the main reservoir of the system (useful capacity of 1112 Mm^3). Figure 34 shows the simulated MROC for each time steps (the blue dots), and associates it to reservoir storage at each time step. Similar plots are produced for each reservoir in the system, to fully model the economic value of water in each point of the Jucar River basin. Then, the storage capacity of each reservoir is divided into ranges where the average MROC is different: this is what is presented in Figure 35.

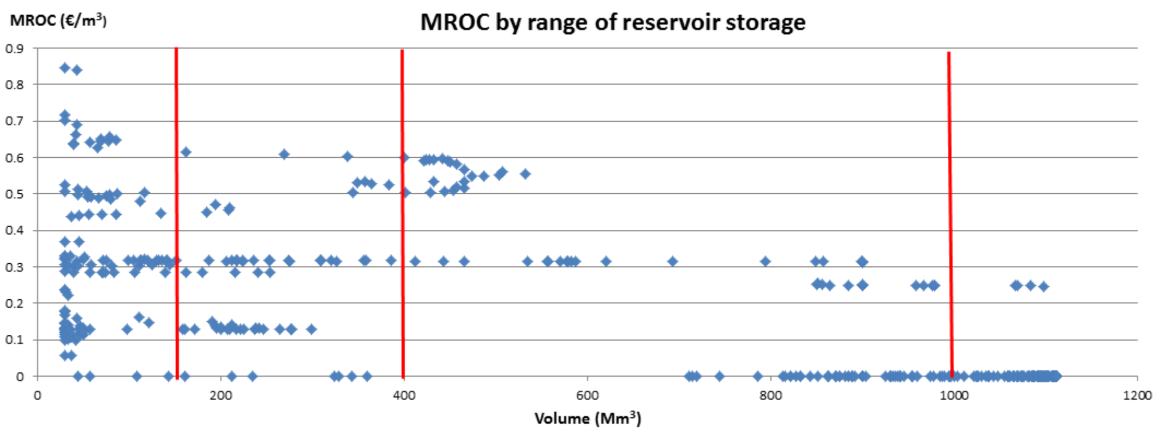


Figure 34. MROC by range of reservoir storage of Alarcón.

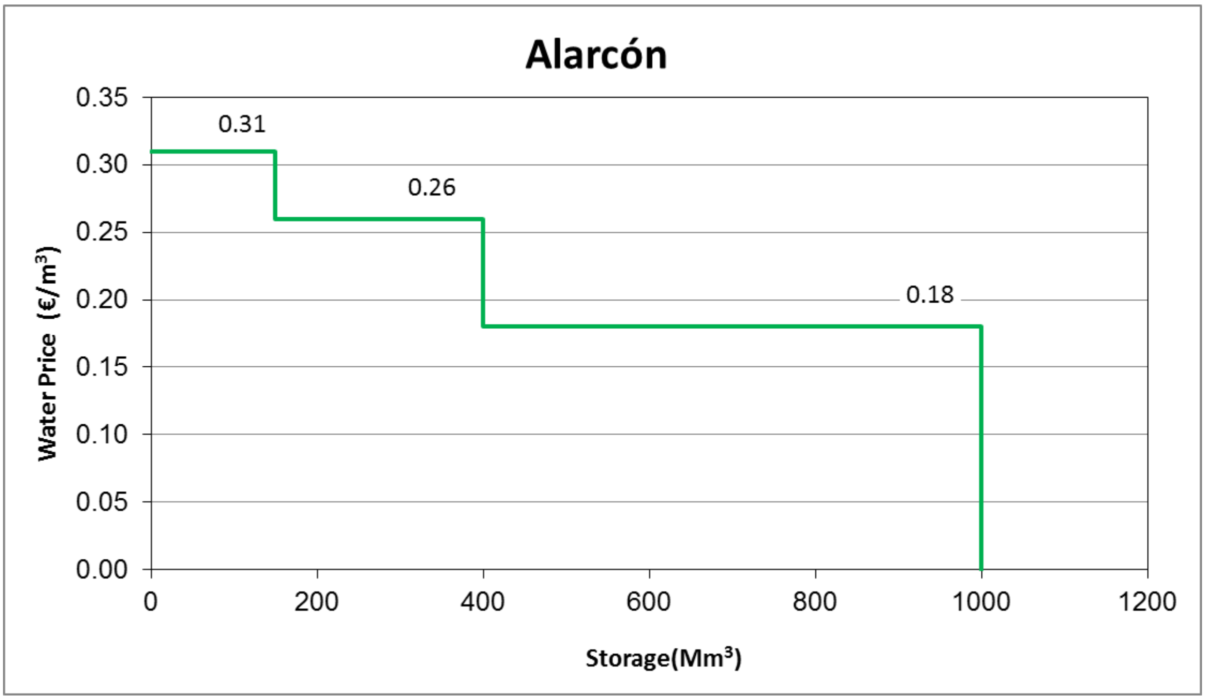


Figure 35. Step pricing function for Alarcón reservoir.

5.3.2 Determining a baseline tariff

Once a MROC has been assigned to each storage range in each reservoir, the methodology proposed in Section 5.2.1 is applied. Its first step is to design the baseline water tariff, getting the values of λ_{1b} and λ_{2b} (€/m^3). Therefore, it is necessary to assess the total costs and the fixed revenue. The total cost has been estimated as 72.14 M.€/year, whilst the volumetric charge cost is estimated as 0.472 €/ m^3.

Table 7 shows the fixed revenue, taking into account service, maintenance and other charges for different meter diameters. The total fixed cost is estimated as 47.91 M.€/ year.

Meters	Service charge (M. €/ year)	Other charges (M. €/year)	Maintenance Charge (M. €/ year)
<15 mm	27.34	4.54	12.58
20 mm	0.82	0.16	0.41
25 mm	0.17	0.04	0.08
30 mm	0.21	0.04	0.09
40 mm	0.26	0.05	0.10
50 mm	0.25	0.05	0.11
65 mm	0.22	0.04	0.07
80 mm	0.08	0.01	0.02
100 mm	0.11	0.02	0.03
TOTAL	29.46	4.94	13.51

Table 7. Fixed income for Valencia (M.€/ year) (source: Aguas de Valencia)

In order to get the volumetric charges for each block,λ_{1b} and λ_{2b} (€/m^3) , the total demand per block has been evaluated. Table 8 shows the total water demand per year at user level, amounting to a total consumption of 46.12 Mm3/year. Assuming an efficiency of 90 %, we estimate the total domestic raw demand to be 51.25 Mm^3/year.

Ranges	Users	m^3 / bi-monthly /user	m^3 / bi-monthly /total	Mm^3/year/total
(0 - 8) m3/bi-monthly	161,895	8	1,295,160	**7.77**
(8- 12) m3/bi-monthly	801,95	12	962,340	**5.77**
(12 - 25) m3/bi-monthly	146,114	25	3,652,850	**21.92**
(25 - 40) m3/bi-monthly	31,304	40	1,252,160	**7.51**
(more than 40) m3/bi-monthly	10,492	50	524,600	**3.15**

Table 8. Total water demand at user level

After that, the baseline water tariff has been obtained by taking a value of λ_1 of 0.44 €/m^3 and then by applying the revenue sufficiency condition. Figure 36 shows the baseline water tariff for Valencia, being the price for the first block 0.44 €/m^3 and the ones corresponding to the second block of 0.56 €/m3. The threshold of the first block, 12 m^3, is due to equity conditions. These results are consistent with the real water tariff.

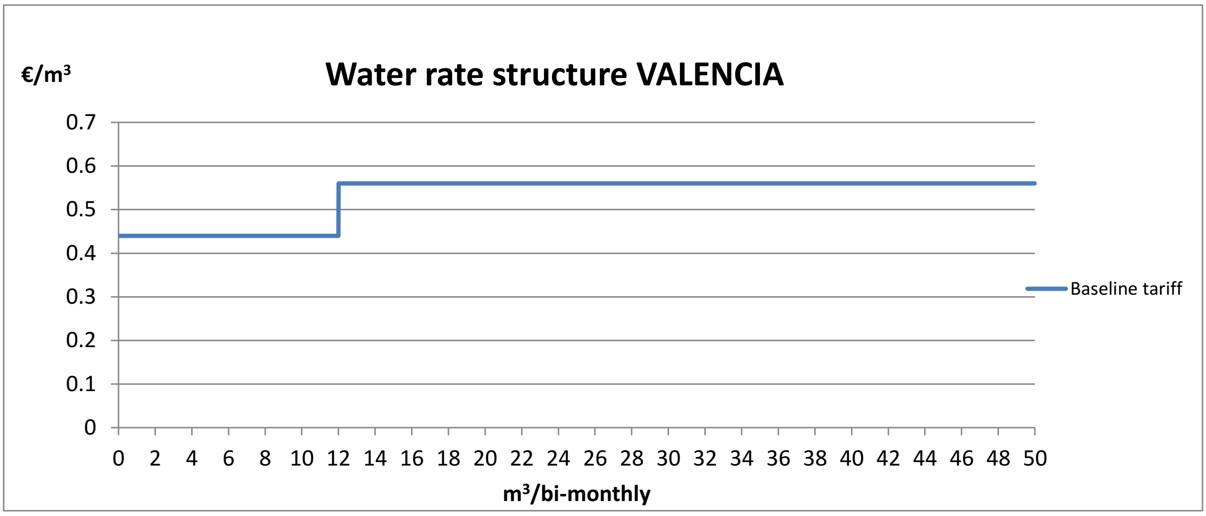


Figure 36. Baseline water tariff.

5.3.3 Scarcity tariff, Approach 1

Once we have obtained the baseline tariff, we can design the new dynamic water tariff considering the two proposed approaches. Regarding the first approach, Figure 37 shows the set of water tariff associated to the different MROC values linked to the storage volumes. In the three cases, the charge of the first block remains constant, whilst the charge of the second block is higher when the volume of the reservoirs is lower.

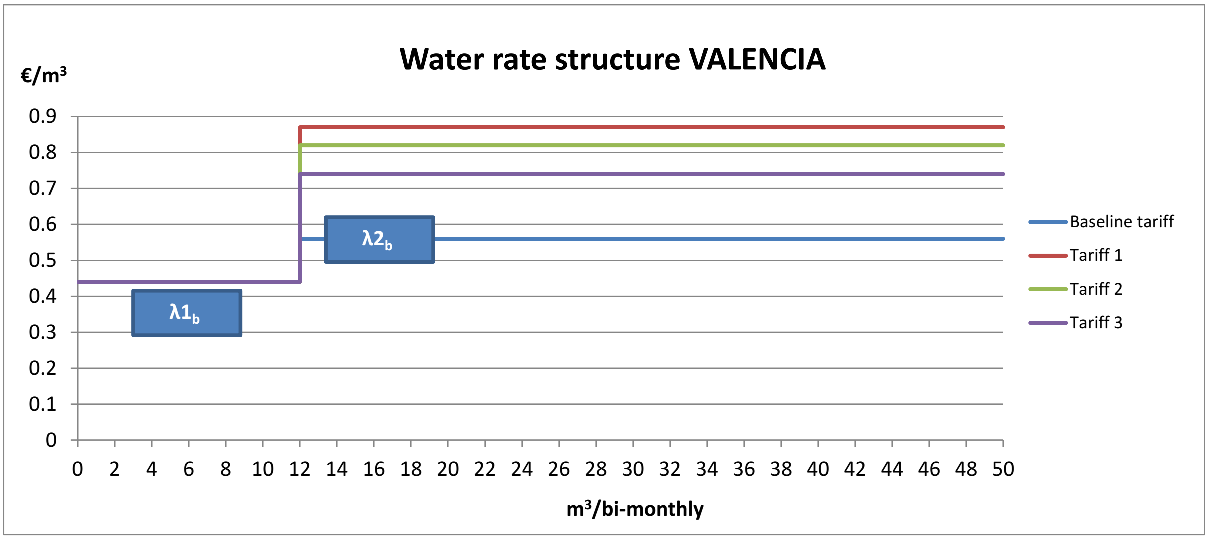


Figure 37. Water rate structure for urban water supply to Valencia. MROC approach (1).

Table 9 presents the new values for λ_2 associated to the different values of MROC. The three tariffs allow reducing the water supplies during the scarcity situations, and producing positive net incomes. As the water tariffs have to meet the revenue sufficiency criterion, the extra income should be re-invested in order to increase the efficiency of the system (e.g. reducing loses of the pipes) and reducing the socio-economic impacts during the scarcity situations.

	MROC (€/m^3)	λ_2 (€/m^3)	% water reduction	Net revenue (M.€ / year)
Tariff 1	0.31	0.87	25%	6.1
Tariff 2	0.26	0.82	21%	5.5
Tariff 3	0.18	0.74	14%	4.4

Table 9. Analysis of water rate structure of Valencia. Theoretical approach (1).

5.3.4 Scarcity tariff, Approach 2

Regarding the second approach, we have adopted the predefined targets of reduction of water that Table 10 shows, linked to different ranges of storage levels (different scarcity situations). Moreover, Figure 38 shows the three water tariff associated to the three saving targets. The results demonstrate the accuracy of applying water tariffs that consider water saving targets, during the scarcity conditions. Moreover, the extra-revenue should be re-invested.

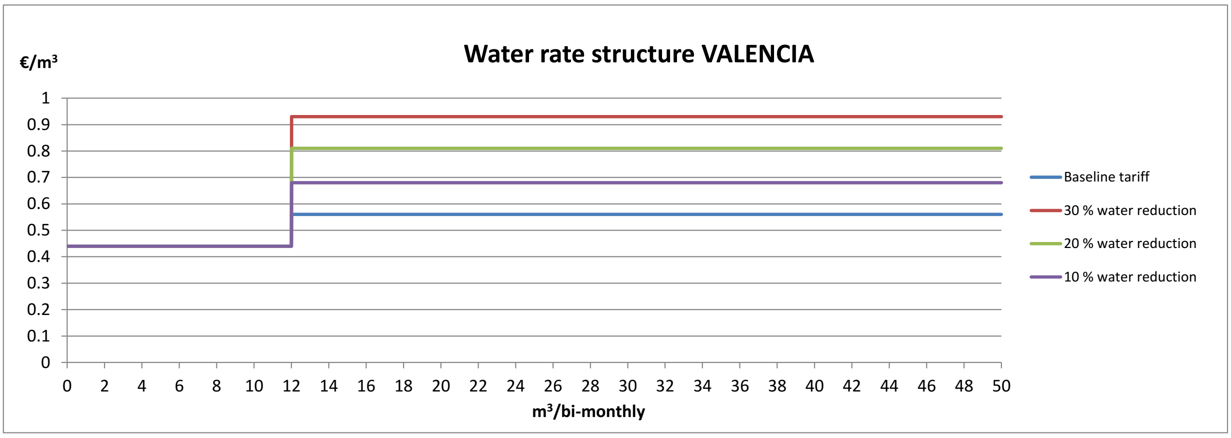


Figure 38. Water rate structure of Valencia. Predefined scenarios of water saving targets (Approach 2).

Scarcity situation	% water saving	λ_2 (€/m^3)	Net revenue (M.€ / year)
Worst	30%	0.93	6.4
Intermediate	20%	0.81	5.42
Regular	10%	0.68	3.28
Good	0	0.56	0

Table 10. Analysis of water rate structure of Valencia. Predefined scenarios of water saving targets (Approach 2).

The proposed methodology allows the design of a dynamic water tariff, considering the role of water pricing as an incentive for an efficient management of water demands during scarcity periods. This is done by setting water prices for different water saving targets or by integrating an MROC component in the tariff, without modifying the reduced price considered for equity reasons and keeping the revenue sufficiency condition. The excess of revenue generated during the scarcity periods (partially compensated by a lower demand) could generate additional resources for increasing water security and reliability.

6. Concluding remarks

6.1 Summary of results

This deliverable established the marginal resources opportunity cost (MROC) of water as a universal indicator of water scarcity and as a basis for establishing dynamic pricing schemes. The design of economically efficient water pricing schemes, and their consequence in terms of price increases and of water savings, both require the use of basin-wide hydro-economic models.

The deliverable proposes two such studies with very different characteristics, in London (UK) and Valencia (Spain). Despite those differences (discussed in Section 0), both demonstrate the potential of dynamic scarcity pricing for reducing consumption while keeping price increases almost entirely in the 0-100% range.

The London case-study looks at future water scarcity (mid-21st century) in a context of population growth, and for different values of the economic parameters necessary for the design of efficient dynamic pricing schemes. It shows both the strengths of scarcity pricing for preserving the Thames' environmental flows during droughts of medium severity but also its limitations in extreme droughts conditions. This highlights that, similar to any policy, dynamic scarcity pricing, despite its potential, should not be considered as a panacea, and should instead be implemented with other interventions aimed at guaranteeing that water supply meets the demand at all times.

The Valencia case-study looks at present water scarcity in a context of a significant share of water use for irrigated agriculture, 83 %, in the Jucar river basin where the city gets most of its water It shows how a dynamic scarcity pricing can be implemented while also complying with the other objectives water tariffs have, such as equity and revenue sufficiency. By indexing tariffs on reservoir levels as a result of hydro-economic modelling, rather than on arbitrary demand reduction target, they also propose a basis for designing tariffs that urban users could accept so they can contribute to lowering scarcity in the area and promoting a more efficient use of water.

6.2 Comparison of the case-studies

The difference between the case-studies are summarised in Table 11.

Characteristic	London case-study	Valencia case-study
Scarcity	In the future	At present
Share of urban use in basin	Dominant	Minor but non-negligible
Storage capacity	Small (A few months)	Large (Interannual)
Price elasticity of demand	Unknown	Already studied
Other users	Environmental flows	Mainly agriculture
Other users' benefits	Rough evaluation	Demand curves available

Table 11. Comparison of the case-studies.

These differences enabled to assess scarcity pricing in different situations:

- Scarcity: investigation of future scarcity enabled to assess the impact of various assumptions on scarcity pricing, and show that while its efficiency was influenced by these parameters' values, its overall relevance did not. Investigation of present scarcity, instead, enabled to present a ready-to-use methodology for tariff design, accounting for traditional objectives of urban water tariffs.

- Share of urban use / Other users: scarcity pricing in a basin where urban use is dominant enables to measure the direct impact of policies on urban flows. In contrast, in a basin where agricultural use dominates, scarcity savings from urban use should be implemented jointly with scarcity savings from other sectors (e.g., irrigated agriculture) to deliver greater levels of basin-wide economic efficiency and water supply reliability.
- Storage capacity: this deliverable demonstrates that the relevance of scarcity pricing does not depend on the drought-buffering capacity of the reservoirs. However, it also shows how different drought-buffering capacities lead to different dynamic pricing schemes: price variations on shorter timescales become relevant as a weaker buffering capacity causes the situation to evolve more quickly.
- Knowledge of the parameters: a better knowledge of the economic parameters needed to design a dynamic scarcity pricing scheme is crucial. Without this, no practical implementation would be possible.

6.3 Recommendations for future work

In a context where smart metering is set to profoundly change the relationship between water utilities and their residential customers, it is normal to assess the impacts of possible changes in the main aspect of that relationship until now – the collection of fees by the water utility. This deliverable is doing just that by proposing one economic indicator – the MROC – and a general methodology – hydro-economic modelling to design scarcity pricing schemes, then assess their basin-wide impacts. It shows the overall potential of such schemes while also highlighting what data is needed for their precise assessment and their practical implementation.

6.3.1 Smart metering and price response

A central assumption in assessing these schemes concerns the demand curve, which describes aggregated consumption levels as a response to price changes. In this work, the price elasticity of demand is taken to be a constant, and the demand curve is derived from this single value. The assumed functional form of the demand curve is also an assumption that can have a huge influence on the results, especially when price increases become large. The comparison of the exponential and linear point expansion methods in the London case-study illustrates the point.

In fact, smart metering, and dynamic pricing, may upend these assumptions for several reasons. First, other ways to engage the customer, for instance those envisioned in the SmartH2O project (consumer engagement portal, gamification, online pricing tool, etc) combined with traditional awareness-rising instruments (e.g. advertisement campaigns), may have a large influence on the price elasticity of demand, and therefore change it from values of the pre-smart metering era. Secondly, repetitive price changes, which are likely to be the norm with dynamic pricing, may affect the price response over time. There already is evidence that for a single price change, there is an adjustment period as the tariff change, so that the price elasticity of demand takes time to converge towards a steady value. Therefore, it seems reasonable to imagine that the price elasticity of demand could be influenced by dynamic pricing.

How might these factors influence the price response in reality? There is no way to tell before the first implementations of dynamic scarcity pricing in a real-world context. A more relevant question would therefore be: how might we ensure that the price response is as effective as possible and does not wane over time? These are questions that cannot be dealt with through hydro-economic modelling, which is the central methodology used in this deliverable, but through behavioural economic and econometric methods. In other words, this is a very complex topic, explored in “D5.4 Experimental economics-based tests of pricing policies”

6.3.2 Towards multi-objective, robust scarcity pricing schemes?

Other data has proved to missing or incomplete in the assessment of demand curves, central to hydro-economic analysis and to the design of economically efficient water pricing schemes. For instance, there is no thorough and precise evaluation of the value of environmental flows in London, including its recreational value for tourism and its impact on property values, in addition to a valuation of ecosystems services. What should be included in that benefit function is itself subject to debate. For instance, should the value added to properties by the proximity to the Thames River in London be factored in an environmental demand curve the same way as protection of biodiversity would be? This goes towards showing that all uses of water cannot easily be put on a single monetary scale. Likewise, the objectives of urban water tariffs such as the ones proposed in the Valencia case-studies may be competing to a certain extent.

In this context, an alternative to *a priori* setting of monetary values to represent different objectives of different users / different objectives of water tariffs is to instead find the dynamic scarcity pricing schemes that perform well regarding several stated objectives. In London, this would represent an indirect valuation of Thames environmental flows: each level of in-stream shortage would correspond to a price of residential water that reflects the MROC given to environmental water in that situation. Higher scarcity prices would reflect a higher premium on protecting the environment, and would indeed protect the environment more, but they may be less socially acceptable. This multi-objective view of tariffs would enable to build economically efficient tariffs that are found according to the explicit preferences of different stakeholders, reinforcing their social acceptability.

Another assumption in the current modelling is the stationary nature of the hydro-economic models used. Demand levels and hydrological supply are both assumed to retain the same probability distribution through time, and this enables to use the available data to design a scarcity tariff. Yet, both supply sources and demand may evolve rapidly under the combined effects of climatic, land-use, social and technological change. In this context, in addition to having a tariff that is economically optimal (i.e. efficient) in what is thought to be the current situation, it seems important in future assessments of scarcity tariffs to test the robustness of these tariffs (and their objectives) to changing conditions.

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