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Institutions and Economics of Water Scarcity and Droughts

Edited by

Julio Berbel, Nazaret M. Montilla-López and Giacomo Giannoccaro

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Cover image courtesy of Nazaret M. Montilla-López.

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About the Editors

Julio Berbel is a full Professor of Agricultural Economics at the University of Córdoba where he is coordinating projects related to environmental management and biotechnology. He holds a Ph.D. in Agricultural Engineering (University of Córdoba) and a Master in Agricultural Economics (University of Manchester). His publications are in the field of water, agricultural economics, agribusiness, and environmental management. He has been a consultant and technical assistant to the European Commission (DG ENV-Water), Ministry of Agriculture and Environment (Spain), and Junta de Andalucía for the implementation of the Water Framework Directive and River Basin Management Plans. He combines hands-on experience in the environmental and agribusiness management sector with academic and scientific involvement. His research focuses on water economics, specifically economic instruments for water governance, including water pricing, water markets, water accounting, and cost-effectiveness analysis. His models for policy analysis are based on mathematical programming at the farm level through multi-attribute utility theory (MAUT) and multi-criteria decision making (MCDM). He also focuses on water use and efficiency, including the rebound effect of investment in water saving, and bioeconomy and the circular economy in food and agricultural sectors.

Nazaret M. Montilla-López has a degree in Agricultural Engineering (specialized in Agricultural Economics), University of Córdoba (Spain), and a Ph.D. in Agricultural Economics (within the Postgraduate Program Agricultural Engineering, Food, Forestry and Sustainable Rural Development). She has six years of experience in water resources management. Her research interests are related to the economic instruments for water governance, with a specific focus on mathematical programming, using specific software called General Algebraic Modelling System (GAMS). She developed her dissertation through a research project called "Designing new water markets for Spain: Assessment as measures for the improvement of efficiency in water use and the adaptation to climate change" (MERCAGUA). The results have been presented in national and international conferences. In 2018, she was awarded the best Ph.D. in Water Economics by Cátedra Aquae de Economía del Agua.

Giacomo Giannoccaro has a B.Sc. in Agricultural Sciences and Technologies from the University of Bari, Italy, and a Ph.D. in Agricultural Economics (Management of Innovation in the Agriculture and Food systems of the Mediterranean Region) from the University of Foggia, Italy. He has 15 years' experience in the economics of water resources. His research interests are related to the economics of agricultural biomass as feedstock, with a specific focus on farmer behavior toward innovation, sustainability, and willingness to change on-farm activities. He exhibits thorough knowledge of farm modeling through linear programming models. His research interests are related to environmental analysis, farmers' attitudes and behaviors through qualitative and quantitative surveying methods (such as focus groups, contingent evaluation, and choice modeling), and analyzing the resulting data using econometric methods. He is routinely working in both Spain and Italy.

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Editorial

Institutions and Economics of Water Scarcity and Droughts

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1. Introduction

Integrated water resources management seeks an efficient blend of all water resources (e.g., fresh surface water, groundwater, reused water, desalinated water) to meet the demands of the full range of water users (e.g., agriculture, municipalities, industry, and e-flows). Water scarcity and droughts already affect many regions of the world and are expected to increase due to climate change and economic growth.

In this Special Issue, 10 peer-reviewed articles have been published that address the questions regarding the economic effects of water scarcity and droughts, management instruments, such as water pricing, water markets, technologies and user-based reallocation, and the strategies to enhance resiliency, adaptation to scarcity and droughts. There is a need to improve the operation of institutions in charge of the allocation and re-allocation of resources when temporal (drought) or structural over-allocation arises.

Water scarcity, droughts and pollution have increased notably in recent decades. A drought is a temporary climatic effect or natural disaster that can occur anywhere and can be short or prolonged. Water scarcity involves a lack of supply relative to potential or current demand that generates conflict between alternative uses of water, especially regarding the requirements of societies, economic sectors, territories and ecosystems. Traditionally, users in water-scarce regions have adapted to dealing with water shortages; however, droughts can greatly increase problems since they are uncertain events and also affect water-abundant regions, with climate change increasing their frequency and severity [1].

Supply-side mechanisms have traditionally been employed to cope with drought by building infrastructure (wells, dams, channels, inter-basin transfers), and recently by including desalinised, brackish, and reclaimed wastewater into the resource mix. Berbel and Esteban [2] study the influence of drought as a catalyst for water policy reform in three developed economies with a Mediterranean climate (Spain, California and Australia), and find that solutions and institutions are trajectory-dependant and grounded in social institutions. Nevertheless, there is a convergence of the type of instruments employed to manage water scarcity and droughts.

In this Special Issue, the reported case studies recount experiences from USA, China, and the European Union (southern Member States). A variety of proposals aimed at tackling droughts and scarcity have been discussed, ranging from economic tools (pricing and insurance) and the increased use of reclaimed wastewater, to reforming the institutional setting (water markets and priority rights). Most of these papers analyses economic instruments and agriculture, but other economic sectors as well as non-market values are also addressed.

2. Papers Contributed

2.1. Demand-Side Policy

The use of demand-side policy is represented by Torres-Bagur, et al. [3]. Tourism activities have been steadily increasing in the last decade, thereby adding competition for water sources. Tourism activities show the highest per-capita water use, and conflict between water users for water re-allocation (generally from agriculture to tourism) arises in regions already affected by water scarcity. In order to make tourist accommodation more sustainable, strategies to promote efficient water use therein can be established. The contribution is focused on sustainable water consumption and resource management in the tourist accommodation industry. The authors conduct a survey on guests staying at campsites, hotels, and rural lodgings in the Muga river basin (Spain) and report that the adoption of water-saving practices is largely influenced by the sociodemographic and motivational features of the guests. The most common tourism is that of the sun and beach sector, which unfortunately reports behaviour of a less sustainable nature. The most relevant finding is that three-quarters of the tourists surveyed declared that they would be willing to reduce their water use subject to economic compensation.

2.2. Governance of Water Rights

Firstly, a proposal for the implementation of guarantee-differentiated water-right entitlements is proposed as an alternative to the current water rights based on the proportional rule, which fails at guaranteeing the water supply as a relevant attribute in the allocation [4]. The argument is based on the allocative efficiency of current rules implemented during drought periods to reallocate the available water resources during a declared drought. An exhaustive review of alternative allocative instruments is presented with an examination of examples. The case of Australia offers the best real example of guarantee-differentiated priority rights that have been in place since 2000. A proposed framework for Spanish irrigation water is presented which includes differentiated tariffs with a lower charge for ordinary rights and higher charges for priority use.

Irrespective of institutional reforms, the establishment of water allocation rules entails transaction costs. The contribution by Loch et al. [5] deals with transaction costs related to transitions between institutions. Drought management institutions in the Po basin (Italy) are investigated by focusing on transaction costs for transitioning drought management institutions towards informal, participatory, and consensus-based approaches (i.e., a Drought Steering Committee). The contribution has found that costs for establishing, coordinating, and managing drought events through informal arrangement have fallen over time as proof of efficient institutional organisation.

Water markets as a way to allocate scarce water resources have long been recognised among institutional reforms in the United States. The role that water markets have and might play in addressing scarcity in the Southwestern United States, namely in Arizona, Texas and California, is studied in Schwabe et al. [6]. The analysis reports the volume and value of water traded on water markets over the last decade (2009–2018) taking spatial, temporal and sectorial features into consideration. The results show that water-right leasing has increased over time, and has dominated the market share in terms of traded volume and value, with farmers being the primary sellers. While new trading frameworks are emerging, as in the case of groundwater banking and storage water rights in California, the water market remains narrow, the explanation for which may lie in the existing transaction costs, the out-of-region ban, and third-party effects. All these issues are recognised among the challenges facing the water market.

2.3. Hydro-Economic Models

Hydro-economic models are a valuable tool for improvement in the understanding of the economic impacts of scarcity and droughts and the evaluation of alternative instruments. An updated review on hydro-economic modelling in the context of climate change is provided in Expósito et al. [7], whose main conclusion points to the limitation of current models in accounting for uncertainties and risks associated with climate change. Future research should deal with such a limitation.

An example of a complete application of the hydro-economic model is given by the analysis that Borrego-Marín et al. [8] include in the Guadalquivir river basin, where the re-allocative effects of water-pricing policy are analysed. Based upon a simple model, the multi-sectoral impacts of water pricing are assessed at the basin scale. The water–demand curve is drawn for domestic, industrial, recreational, and agricultural uses. According to the results of the model, the irrigation water price needs to be increased three-fold to force savings and re-allocation from agriculture to sectors of a more profitable nature.

The Guadalquivir River basin example shows that water pricing is a limited instrument for water reallocation, thereby confirming the general findings regarding this instrument related to WFD implementation in the EU [9]. Institutional reforms and changes in water rights management are discussed in the subsequent set of three contributions.

2.4. Supply-Side Enhancement

Analysis of causes for changes in water areas of the Baiyangdian Lake (China) are investigated in Wang, et al. [10]. By combining Landsat images with hydrological and climatic ground data, the extraction of surface water from 1984 to 2018 is studied. The lake area has been affected largely by human activities, and, to a lesser extent, by climate change. The development of artificial water diversion projects for lake replenishment from neighbouring basins seems to have succeeded in reaching a good ecological status. However, such solutions are no longer possible.

The remaining contributions deal with the enhancement of water supply. Firstly, water-supply reliability and its economic impacts are analysed by Sjöstrand, et al. [11]. In this research, the contribution studies the failure in water provision to domestic users and proposes a risk assessment method. The approach is illustrated on the island of Gotland (Sweden), since it is the country's most water-scarce area, where the solution proposed involves an increase in surface water extraction as the most cost-efficient risk-reduction alternative.

Although reclaimed wastewater is envisioned as a reliable water supply, its use remains undeveloped. Barriers and opportunities for reclaimed wastewater use for agriculture in Europe are surveyed in Mesa-Pérez and Berbel [12]. The paper aimed to explore the impact of the recently approved Regulation EU-2020/741 "Minimum requirements for water reuse in agriculture". The perception given by almost a hundred interviewees regarding key actors across eight European Member States is investigated. Two main groups of countries are found: (a) those concerned about the cost of implementing, distributing, and storing reclaimed water; (b) countries where social and governance issues are the most pressing aspects.

2.5. Innovative Economic Instruments

Finally, the research by Guerrero-Baena and Gómez-Limón [13] regarding insurance for ensuring irrigation water supply is discussed. A new index-based drought insurance scheme for irrigation, linked to the stock of water available in reservoirs, is proposed. An illustrative example is reported for the Guadalquivir River Basin, located in southern Spain. The main conclusion is that insurance schemes against irrigation shortage may be an available instrument in the future, but further research is required to develop a commercially affordable service.

3. Concluding Remarks

The contributions to this Special Issue highlight the key aspects of institutions that may tackle not only the increasing water scarcity in many regions of the world but also the increasing frequency and impact of droughts in economic and natural systems. Some of the papers analyse critical issues, such as the state and future trends of water markets, the estimation of transaction costs when dealing with drought management, the use of new instruments, such as insurance and water-right entitlements that include water security, water-pricing effects on a whole basin, and intra- and inter-sectorial re-allocation. Furthermore, the important issue regarding non-conventional water supply and the governance of the new resources also features as the topic of some of the contributions.

To conclude, this issue provides an in-depth revision of the main aspects of institutions and instruments available for the management of droughts and scarcity governance, and has opened a new field of research in certain emerging innovative instruments.

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Article

Understanding the Key Factors That Influence Efficient Water-Saving Practices among Tourists: A Mediterranean Case Study

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Abstract: The future of tourism activity is dependent on its ability to adapt to the effects of climate change. One of the most notable effects in the Mediterranean area will be water shortages in a scenario marked by increasing demand for this resource. While this situation will affect numerous economic sectors, it will have a severe impact on the tourism industry, which relies heavily on water. The aim of this study was to analyze water-saving practices among guests at campsites, hotels, and rural lodgings in the Muga river basin and investigate the factors that influence these practices. We conducted 752 surveys and found that differences in practices were influenced by reason for stay, type of accommodation, and geographic origin. A greater understanding of how sociodemographic and motivational characteristics influence water-saving behavior by guests at different types of tourist accommodation is essential for designing targeted strategies for improving environmental awareness and water-saving habits.

Keywords: good water practices; tourist accommodation; tourist; Mediterranean; water scarcity

1. Introduction

Water is an essential resource for the tourism industry. It is needed both for human consumption and to support key infrastructure and facilities, such as swimming pools, spas, and golf courses. Access to sufficient supplies of quality water is a growing concern in the industry, particularly in destinations prone to shortages due to the effects of climate change and growing demand [1]. Worldwide supply projections in the short and medium term are alarming, and Catalonia is no exception. A report published by the Catalan Government in 2016 [2] forecast a reduction in water availability of approximately 11% between 2015 and 2021 and 17.8% between 2015 and 2051. In addition, findings from the European Life MEDACC project [3] showed that summer rainfall in the Muga river basin (where this case study was conducted) decreased by approximately 60% between 1973 and 2013 and are expected to decrease by a further 7.5% by 2050. The projected reductions for spring and autumn rainfall are even higher, at 11.5% and 15.1%. This continued decline in rainfall over the decades has led to reductions in river flows. In the case of the Muga river basin, headwater flow decreased by 50% between 1973 and 2013 and is expected to decrease by an additional 20% by 2050. This worrying scenario, added to growing pressure on water resources from numerous economic sectors, calls for urgent action from all quarters, in particular the tourism industry and other key stakeholders with competing interests, such as industry, agriculture, and conservation groups.

The various players in the tourism industry need to adapt to the changing scenario and implement appropriate water-saving measures. Building on previous work by our group on perceptions of climate change among tourist accommodation establishment managers and incentives and barriers to the implementation of water-saving measures in hotels in the Muga river basin, in this article we investigate factors that explain variations in water-saving habits among guests staying at campsites, hotels, and rural lodgings in the area. Information of this kind is essential for guiding the design of strategies aimed at increasing environmental awareness and fostering good water-saving habits among guests at hotels and other types of tourist accommodation.

The rest of this article is structured into seven sections. We first discuss the theoretical framework underlying our study and then describe the methodology and study area. Next, we analyze and discuss our results in light of the existing literature and close with a series of conclusions and practical recommendations for promoting best practices in water consumption and management.

2. Theoretical Framework

The tourism industry is one of the largest and fastest-growing industries worldwide, although it remains to be seen how it is affected by the novel coronavirus 2019 pandemic (COVID-19), at least in the short term. According to the World Tourism Organization of the United Nations (UNWTO, Madrid, Spain), international tourist arrivals worldwide grew by 5% in 2018 to reach a total of 1400 million [4], and this growth has an obvious impact on the accommodation sector [5]. International tourism, however, is highly sensitive to safety and security issues, as clearly evidenced by the COVID-19 pandemic. The industry has been one of the hardest hit sectors since the outbreak and the introduction of worldwide lockdown measures involving travel bans, closing of borders, confinement measures, and quarantine periods [6]. According to estimates by the UNWTO, international arrivals this year will fall by approximately 20% to 30% compared to 2019 [7], and one can expect Mediterranean countries with high caseloads, such as Italy and Spain, to be particularly hard hit. Restructuring strategies designed to build resilience to future crises are necessary, particularly in major tourist destinations.

Nonetheless, current efforts to combat the COVID-19 pandemic must not lead us to lose sight of other structural crises threatening the future of tourism—and humanity. Climate change [6] and overexploitation of natural resources [8] will continue to be global challenges with potentially devastating consequences. Fortunately, recent polls conducted by Ipsos MORI (Ipsos MORI, London, UK) in 14 countries, including Spain, the United States, Canada, and China, have shown that 70% of the population continues to consider climate change and environmental problems to be as serious a crisis as COVID-19 and that 65% of respondents believed that the fight against climate change should be prioritized in post-coronavirus economic recovery [9]. It is, therefore, more necessary than ever for key stakeholders to continue to take appropriate environmental decisions to facilitate the implementation of water-conservation measures that will protect both present and future availability [10–12]. A growing number of hoteliers and other accommodation owners are adopting measures aimed at increasing the sustainability of their business, while at the same time improving corporate image [13,14] and lowering operating costs [15–18]. Their actions can also help increase customer satisfaction and build loyalty [19] among a clientele that is increasingly aware of the detrimental effects of human activity on the environment. Recent years have witnessed the emergence of a new type of tourism characterized by visitors who prefer to stay at green establishments and who are willing to pay extra to do so [20]. That said, a report by the U.S. Travel Association (US Travel Association, Washington, USA) in 2009 [21] found that just 9% of clients were willing to pay more for green travel options. Without the engagement of guests, strategies adopted to promote efficient water use at tourist accommodation establishments will lose much of their effectiveness [22]. While it is true that the general public is increasingly aware of the importance of efficient water use, particularly since the turn of the millennium, considerable differences have been observed between what people do when at home and when on holiday, and, as reported by Barberán, Egea, Gracia-de-Rentería, and Salvador [23], Deyà and Tirado [24], Gatt and Schranz [25], and Gössling [26], the differences are even more striking when behaviors at accommodation establishments are analyzed.

A greater understanding of how tourists use water is essential for guiding the design of effective water-saving policies and measures. Water-saving habits in hotels and other establishments have been analyzed in numerous studies. Examples of good habits are turning off the tap while washing your hands or brushing your teeth, turning off the shower while soaping, using a bucket to collect water as it is heating up, and choosing between the reduced- or full-flush options in dual-flush toilets [27,28]. Performance of these actions is closely linked to guest awareness of the need to save water, although the relationship is not always linear [29–31], some authors highlighted the importance of identifying what differentiated guests in terms of good and bad water-saving practices, as this would allow policy makers to more accurately identify target groups for awareness campaigns. Potential differentiating factors identified in the few studies conducted to date include sex, age, geographic origin, and level of education.

In a review of how sociodemographic characteristics influence or explain environmental behaviors, Diamantopoulos, Schlegelmilch, Sinkovics, and Bohlen [32] found that while men were more knowledgeable about environmental practices, women were more aware of and concerned about environmental problems and also more willing to engage in water- and energy-saving practices. Similar findings were reported by De Urioste-Stone, Le, Scaccia, and Wilkins [33] and Han et al. [19], who found that women and young people were more concerned about climate change and future water supply problems and also more proactive in their responses.

Gabarda-Mallorquí et al. [29] classified 648 guests surveyed at a hotel in Lloret de Mar (Girona, Spain) according to their level of environmental awareness and proactivity in terms of saving water and found that differences could be explained by age, sex, geographic origin, and level of education. In Greece, Dimara et al. [20] conducted 1304 online surveys to analyze factors that influenced guest participation in hotel towel reuse programs. They found that young guests, guests who had paid more for their stay, and guests who stayed for longer engaged in better environmental practices and were more willing to reuse towels. Wang et al. [34] found that reason for travel might also explain variations in behavior, as they found visitors at a natural park in Taiwan to be sensitive to the importance of water conservation. In particular, they found that visitors who showed the greatest proactivity in this regard were more knowledgeable about the negative effects of climate change on the landscape. In brief, there is growing consensus among key tourism stakeholders, including tourists, on the need to incorporate sustainable practices in this sector [35,36]. Sustainable water consumption and resource management are particularly important in the tourist accommodation industry [37] if we are to reduce the impact of tourism on our natural environment and ecosystems and in particular safeguard increasingly vulnerable water resources for future generations [38]. For this to occur, multilevel strategies incorporating local, regional, national, and global perspectives and involving all relevant stakeholders are needed.

3. Materials and Methods

We conducted a survey of guests staying at campsites, hotels, and rural lodgings in the Muga river basin to analyze their water-saving habits and investigate associations with sociodemographic and motivational characteristics.

To design the survey, we reviewed the literature to identify key factors associated with water consumption habits among tourists. The information retrieved was used to create a questionnaire validated by members of the Research Group on Water, Territory, Tourism, and Sustainability (GRATTS) at the Autonomous University of Barcelona and the University of Girona. The questionnaire contained 13 closed-ended questions, three sets of items rated on a 5-point Likert scale, and five open-ended questions. It was divided into four sections: (1) guest profile, (2) evaluation of water quality and resources at the establishment, (3) water-saving practices, and (4) general aspects of water consumption and climate change. Considering the diverse geographic origin of visitors to the study area, we prepared the questionnaire in four languages—Catalan, Spanish, English, and French—to avoid possible misinterpretations.

The survey was conducted on-site by interviewing guests at campsites, hotels, and rural lodgings in the Muga river basin. Eligible establishments were identified, and the majority were contacted by email and telephone to arrange suitable times for conducting the surveys. Nineteen establishments (five campsites, ten hotels, and four rural lodgings) agreed for us to survey their guests (Table 1). We first interviewed the manager of each establishment and then surveyed the guests. The surveys were carried out in a public area (e.g., hotel foyer) at each establishment. We conducted 752 surveys; of these, 726 were validated for use in this study as they contained answers to all the questions of interest. The surveys were carried out in 2018, in the months of June (4.8%), July (38.6%), August (42.8%), and September (13.8%). They were therefore carried out during peak business months, when all establishments are open and at full or near-full capacity.

Table 1. Main characteristics of tourist accommodation establishments where guests were surveyed.

		Total Number in Study Area	Number of Establishments Where Guest Interviews Were Held
Hotels	**Category**		
	*	14	2
	**	32	5
	***	48	2
	****	21	1
	*****	2	0
	Total	117	10
	Location		
	Inland	27	2
	Cities	19	2
	Coast	71	6
	Total	117	10
	Number of beds	11,231	531
Campsites	**Category**		
	1 (4-star campsite)	4	2
	2 (3-star campsite)	7	2
	3 (1–2-star campsite)	2	1
	Total	14	5
	Location		
	Inland	4	1
	Coast	10	4
	Total	14	5
	Number of places	10,890	4653
Rural lodgings	**Category**		
	Not categorized	86	4
	Location		
	Inland	82	4
	Coast	4	0
	Total	86	4
	Number of beds	996	53

The analysis of data collected during the field work consisted of different stages. Principal component analysis was applied to data from the first section of the questionnaire (which included aspects such as level of education, age, geographic origin, and main reason for stay) and to four items from the third section:

- I turn off the tap when brushing my teeth.
- I turn off the shower when I am soaping.
- I distinguish between the small and large buttons when flushing the toilet (dual-flush system).
- I use shower water sparingly.

We chose these four questions because, unlike questions on towel and bed linen reuse, we considered they would be applicable to all hotels, rural lodgings, and campsites.

Respondents were asked to indicate how often they applied the four water-saving measures on a 5-point Likert scale, where 1 indicated never; 2, almost never; 3, sometimes; 4, nearly always; and 5, always.

Principal component analysis yielded a single factor, which, together with confirmation of internal consistency using Cronbach's alpha test (>0.7), indicated homogeneous water-saving behavior by individual respondents. Using the ratings assigned to each item, we calculated a mean score (from 1 to 5) to reflect each guest's water-saving behavior during their stay.

The Kruskal-Wallis test [39] was used then to analyze associations between water-saving habits (mean scores from Section 3 of the questionnaire) and guest characteristics (age, sex, country of origin, choice of accommodation, location, and main reason for stay). This test is used to detect statistically significant differences between groups. Statistical significance was established at a p-value of less than 0.05.

We also analyzed answers to two yes/no questions from Section 3 of the questionnaire: (1) Would you would be willing to reduce your water consumption in return for a discount on your stay or another incentive? and (2) Would you would be willing to pay a supplement to be used by the establishment to improve its water-saving measures?

4. Study Area

The Muga river basin is located in the extreme north-east of the Iberian Peninsula, on the border with France. It has 52 municipalities: 46 inland villages or towns, two coastal towns, and two inland cities. Each area attracts a different type of tourist. The coast attracts beach holidaymakers, the inland, more rural area, attracts nature lovers and visitors interested in outdoor pursuits, while the two cities, Figueres (the capital of the region) and La Jonquera (a border town), attract business travelers and urban/cultural tourists (Figure 1).

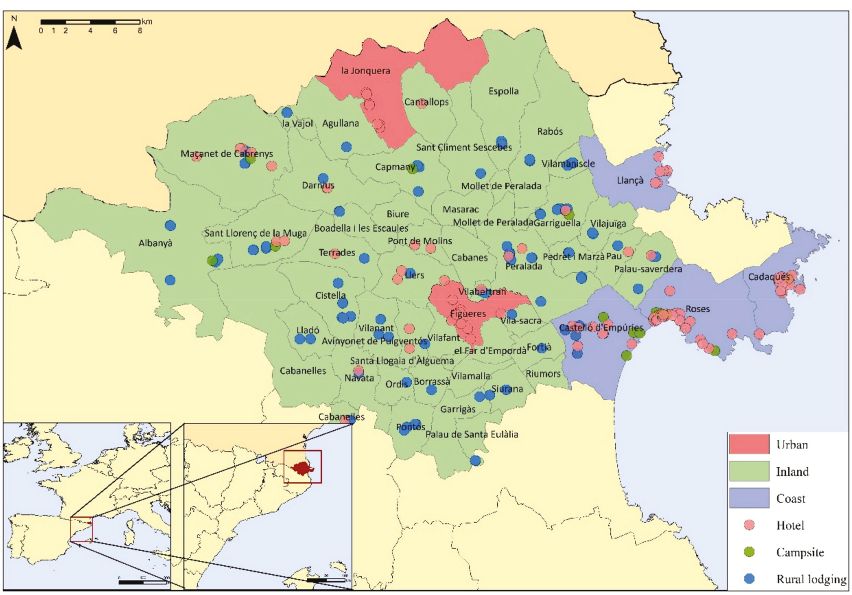

Figure 1. Location of tourist accommodation establishments in the Muga river basin by type.

The Muga river basin has 117 hotels, 86 rural lodgings, and 14 campsites, which together offer approximately 23,000 beds. The bulk of accommodation (80%) is located on the coast. Rural properties offer 15% of all beds, while city hotels offer 5%. The number of visitors has increased continuously since 2012, reversing a 4-year downturn caused by the economic crisis. According to official estimates based on data from tourist occupation surveys of hotels, campsites, and rural lodgings by the Spanish National Institute of Statistics (INE, Madrid, Spain), over 1 million overnight stays were recorded in the area for 2019 [40].

As mentioned, the Muga river basin is particularly vulnerable to water shortages as a result of climate change and rising demand from the tourist industry and other sectors of the local economy [1,2]. The main sources of water are the Darnius-Boadella reservoir and groundwater water extracted by wells. Figueres, the regional capital and home to the largest population in the area, in addition to several coastal towns are supplied directly by the Darnius-Boadella reservoir, while Peralada and Castelló d'Empúries have their own wells, which also supply other inland towns and villages. Demand for water from the reservoir has grown in recent years due to increasing groundwater nitrate pollution in many parts of the area. The response has been to create new connections to the reservoir to guarantee sufficient supplies of water fit for human consumption [41,42].

It is not surprising thus that conflicts between different sectors with competing interests have increased in both intensity and frequency in recent decades. Ventura, Ribas, and Saurí [43] reported 22 such episodes between 1980 and 1999, which corresponds to approximately one episode a year. In 1983, for example, the level of the Darnius-Boadella reservoir fell to just 25% of its total capacity, generating social alarm that led to an increase rather than a decrease in consumption due to fears of restrictions. The situation also generated additional tensions in 1998, when the level dropped to just 8.75% [43].

The main confrontations involving the tourist industry are with the agricultural sector and conservationist groups. Agricultural use accounts for approximately 70% of water use in the region [44], and water is required by law to maintain the river's environmental flows and conserve the coastal marshes in the Aiguamolls de l'Empordà Natural Park, which is an IUCN Category V (Gland, Switzerland) protected area and a member of the Ramsar International Network of Protected Wetland Sites. Tourism, however, has a greater social impact, as it is a key driver of economic growth and job creation. Conflicts between sectors vying for their share of water are more likely in times of scarcity. In 1984, for example, groundwater supplies to the tourist towns of Roses, Castelló d'Empúries, and Cadaqués, dropped dramatically, leading to what became known as the "water well war" [45]. The most recent conflict occurred in 2007 and 2008, sparked by the longest drought recorded in 70 years [46,47]. The Catalan Government took action by introducing a "drought decree" (April 17, 2007) to mitigate the effects of the fast-declining supply of water. The decree remained in force until early 2009 (January 13), when the last of Catalonia's inland river basins (precisely the Muga river basin) emerged from the state of emergency after more than a year without heavy rainfall at the headwaters of the river and with increasingly low reservoir levels and rising social alarm [48–50].

The Muga river basin remains vulnerable to the effects of climate change, and in a scenario marked by increasing demands and decreasing supplies, in part due to the effects of climate change in this area of the Mediterranean, new conflicts are likely to occur if appropriate water conservation and management measures are not taken by each and every one of the sectors that depend on this scarce resource.

5. Results

Of the 752 surveys conducted (Table 2) (726 of which were validated for this study), 53.6% were answered by women and 46.4% by men. The respondents were mostly aged between 26 and 40 years (35.4%) and 41 and 55 years (31%). A majority of respondents had a university education (38.6%), 22.9% had completed vocational training, 17.4% had attended secondary school, and 14.5% had completed upper secondary school education (preparation for university) (14.5%). Over half of the visitors were from Europe (52.4%). French tourists were particularly common, which is to be expected given the proximity of the study area to France. The second largest group of visitors (22%) was from Barcelona city and metropolitan areas. Overall, 73.3% of tourists were from a Mediterranean country. The main reason mentioned for coming to the area was a beach holiday (47.1%), followed by nature tourism (27.7%) and urban/cultural tourism (10%). Less common reasons were sport (7.2%), business (2.4%), events (e.g., concerts, festivals) (2%) and youth and/or school tourism (1.7%). Campsites were chosen by 66.9% of tourists, hotels by 28.5%, and rural lodgings by 4.7%. The most popular area was the coast, which received 59.2% of all visitors, followed by inland areas (34.2%) and cities (6.6%).

Table 2. Main characteristics of guests at the tourist accommodation establishments surveyed.

		n	%
Sex	Male	349	46.4
	Female	403	53.6
Age	<25	95	12.6
	26–40	266	35.4
	41–55	233	31.0
	56–65	89	11.8
	>65	43	5.7
	Unknown	26	3.5
Level of Education	No schooling	6	0.8
	Primary education	39	5.2
	Secondary education	131	17.4
	Upper secondary education (university preparation)	109	14.5
	Vocational training	172	22.9
	University education	290	38.6
	Unknown	5	0.7
Geographic Origin (I)	Girona province	57	7.6
	Barcelona and surrounding metropolitan area	166	22.1
	Rest of Catalonia	54	7.2
	Rest of Spain	66	8.8
	Rest of Europe	394	52.4
	Rest of the world	15	2.0
Geographic Origin (II)	Mediterranean country	551	73.3
	Non-Mediterranean country	201	26.7
Main Reason for Stay	Business	18	2.4
	Sport	54	7.2
	Nature holiday	208	27.7
	Beach holiday	354	47.1
	Urban/cultural holiday	75	10.0
	Events	15	2.0
	Youth and/or school trip	13	1.7
	Unknown	15	2.0
Type of Accommodation	Hotel	214	28.5
	Campsite	503	66.9
	Rural lodging	35	4.7
Location	Coast	445	59.2
	Urban	50	6.6
	Inland	257	34.2

Principal component analysis of data showing the frequency with which guests engage in water-saving practices at the campsites, hotels, and rural lodgings analyzed showed a single component, indicating that individual ratings given to each of the four items on water habits were related. In other words, a guest who turns off the tap while brushing their teeth will also turn off the shower while soaping. Cronbach's alpha for internal consistency was 0.758, indicating that it was possible to calculate a mean score (1–5) for each respondent. The scores had the same significance as the scores on the Likert scale, where 1 corresponded to never (e.g., I never turn of the water while brushing my teeth) and 5 to always ("I always turn it off"). The distribution of scores is shown in Figure 2.

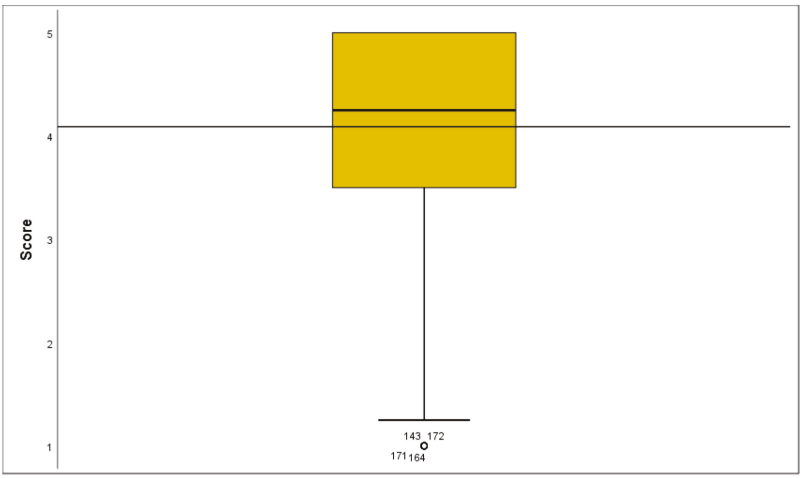

Figure 2. Frequency with which guests engage in water-saving habits on a scale of 1 (never) to 5 (always).

Approximately 75% of the respondents indicated that they always or nearly always turned off the tap while brushing their teeth or showering and that they used the dual-flush system and water sparingly in the shower with the same frequency. Just over half of the guests (55%) stated that they would be willing to pay a supplement to be invested by the establishment in water-saving measures and 73.2% said that they would be willing to reduce their water consumption in return for a discount or other incentive.

Despite these positive results, approximately one in four guests never, hardly ever, or only sometimes engaged in good water-saving practices during their stay. Understanding why can provide important information to guide strategies targeting guests with the worst water-saving habits.

5.1. Factors That Explain Good Water-Saving Practices

Analysis of the association between water-saving behavior and guest profile characteristics revealed significant differences between different types of guests. The Kruskal-Wallis test showed insignificant differences for sex, age, and level of education ($p > 0.05$), but significant differences for main reason for stay, type of accommodation, and geographic origin. These factors are analyzed in the next section.

5.1.1. Main Reason for Stay

Nature tourists were significantly more likely to frequently engage in good water practices than other types of tourists ($p < 0.05$) (Figure 3). Visitors who had come to the area to attend an event had the worst water-saving habits.

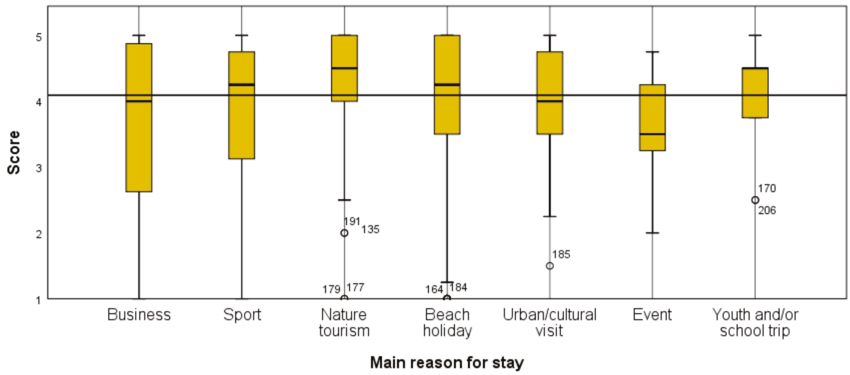

Figure 3. Frequency with which guests engage in water-saving practices by main reason for stay on a scale of 1 (never) to 5 (always).

Over 50% of nature, sport, beach, and youth tourists had above-average scores, while over 50% of business travelers and urban/cultural and event tourists had below-average scores. Nature tourists thus showed the greatest awareness of the importance of saving water while on holidays. Business travelers and cultural/urban tourists had the shortest stays and the worst water-saving habits. Almost 25% of business travelers scored lower than 3, indicating that they never or hardly ever engaged in good water-saving practices. This contrasts sharply with the data for nature tourists, 75% of whom always or nearly always engaged in good practices. Efforts to promote water-saving practices in this case should thus preferentially target business travelers, as well as urban/cultural and event tourists.

5.1.2. Type of Accommodation

Campsite and rural lodging guests generally had better water-saving habits than hotel guests (Figure 4). Sixty percent of campsite guests always or nearly always engaged in good water-saving practices. The mean scores by type of establishment were 4.23 for rural lodgings, 4.15 for campsites, and 3.93 for hotels. The behavior of rural lodging guests was highly consistent, with 100% of those surveyed scoring over 3.6. Behavior at hotels was more heterogeneous. Although half of the respondents scored higher than 4, approximately 30% scored 1 or 2.

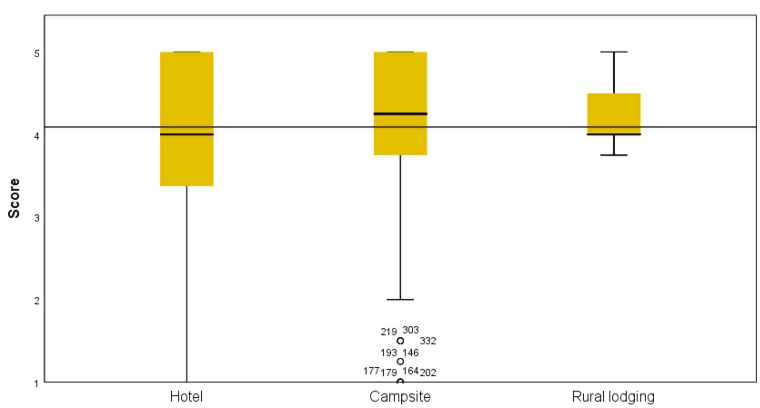

Figure 4. Frequency with which guests engage in water-saving practices by type of accommodation on a scale of 1 (never) to 5 (always).

The profiles of guests staying at campsites and rural lodgings were quite similar, with no significant differences observed for age or geographic origin ($p > 0.05$). Nevertheless, campsite guests stayed for an average of 12.62 nights compared to just 3.56 nights for hotel guests. Number of overnight stays could thus be an interesting factor to explore in future studies, as campsites had the largest proportion of tourists with good water-saving practices.

5.1.3. Geographic Origin

Geographic origin was not significantly associated with water-saving habits in the overall sample, but it was a significant factor in the case of hotel guests ($p < 0.05$).

Specifically, hotel guests from non-Mediterranean countries had worse water-saving habits than Mediterranean guests (Figure 5). Approximately 60% of non-Mediterranean guests scored 3 or lower, indicating that they never, hardly ever, or only sometimes engaged in good water-saving practices. In other words, just 40% of hotel guests from a non-Mediterranean country always or nearly always engaged in good practices, identifying thus a segment to target in good water practice campaigns.

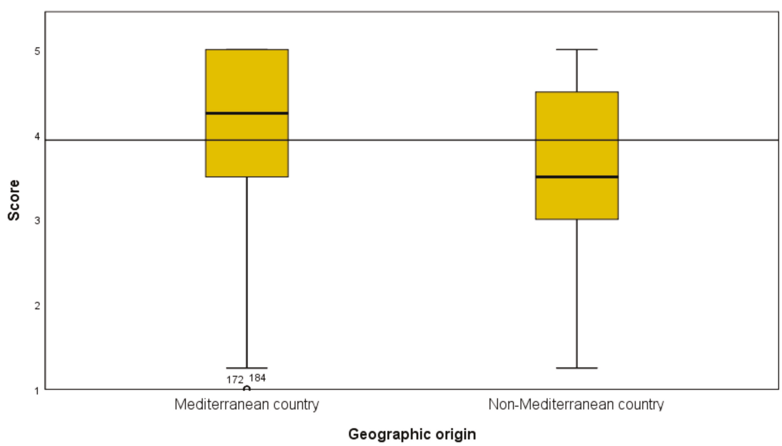

Figure 5. Frequency with which guests engage in water-saving practices by geographic origin on a scale of 1 (never) to 5 (always).

6. Discussion

Surveying of tourists showed high rates of good water-saving practices at accommodation establishments while on holiday. In this case, in the study of water-saving practices by tourists visiting the Muga river basin in north-east Spain, 75% of hotel, campsite, and rural lodging guests reported that they always or nearly always engaged in good water-saving practices. This rate is higher than that reported by Weissenberg, Redington, and Kutyla [51] in the USA, where approximately 60% to 65% of travelers stated that they always or frequently engaged in good water-saving habits, although it should be noted that all those surveyed were business travelers staying at hotels. Despite the high proportion of tourists with good water-saving habits in our study, we were able to identify several factors that explained variations in behavior.

Interestingly, several sociodemographic factors that have been found to be significantly associated with good water-saving practices in previous studies, namely, sex, age, and level of education, were not significant in our study.

Diamantopoulos et al. [32], for example, found that women significantly more likely to engage in good environmental practices than men. Han et al. [19], in turn, found that women were more willing than men to pay extra to stay at a green hotel. In agreement with the findings of other studies, such as a study predicting people's intentions to save water [52], we observed no significant differences between the water-saving practices of men and women.

Gabarda-Mallorquí et al. [29] found that water-saving practices varied according to level of education, with better practices observed in hotel guests with higher levels of education. They also found that older guests were more environmentally aware and willing to save water. Clark and Finley [53] also reported that older household members were more likely to use water sparingly, something they attributed to their having experienced water shortages in the past. Dimara, Manganari, and Skuras [20], by contrast, found quite the opposite in a study of towel reuse programs at hotels, with younger guests more willing to participate in these programs.

Reason for travel was a significant factor in our series, with nature tourists engaging most frequently in good water-saving practices than other types of tourists. This observation corroborates findings by Wang, Lin, Lu, and Lee [34], who reported that tourists who mentioned contact with nature as their main reason for travel tended to have a higher level of awareness about environmental problems, including water scarcity. In our series, good water-saving habits were least common in event tourists and business travelers, most of whom chose to stay at a hotel. This observation is in full agreement with findings by White and Hugues [54] for festival attendees in the United Kingdom.

We also observed better habits among hotel guests from Mediterranean countries. This could be because guests from countries with recurrent drought and similar water scarcity problems might be more environmentally aware and already employ good practices in their home country. As indicated by Gabarda-Mallorquí et al. [29], geographic origin should thus be taken into account when designing water-saving measures for tourist accommodation establishments. Dimara et al. [20], by contrast, in a survey of 1304 tourists in Greece, found that willingness to pay extra for towel reuse was not associated with geographic origin of the tourists surveyed.

7. Conclusions

Water-saving practices at different types of tourist accommodation establishments vary according to guest profile and variations can be explained by several sociodemographic and motivational factors. Reason for travel was one of the main factors that explained differences in water-saving practices while on holiday. Nature tourists were significantly more proactive when it came to using water sparingly than those traveling for other reasons, such as beach tourists, business travelers, and urban/cultural and event tourists. This observation suggests that tourists seeking contact with nature and interested in outdoor pursuits in the natural environment are more environmentally aware. Most of these tourists stayed at campsites or rural lodgings. Tourists with less contact with the outdoors (business travelers, cultural/urban tourists, and event tourists), by contrast, has the worst water-saving habits, and most of them were staying at hotels in the cities of Figueres and La Jonquera.

Similar behaviors were observed among campsite guests and among rural lodging guests, contrasting with the situation of hotel guests, whose behavior was more heterogeneous. Hotel guests from non-Mediterranean countries engaged less frequently in good water-saving practices than their Mediterranean counterparts. Living in a country with similar and possibly even worse drought and water shortage problems, which is the case of most countries in the Mediterranean basin, is thus likely to have a significant influence on awareness of the importance of water and proactivity in relation to the careful use of water while traveling.

Identification of a significant proportion of tourists who did not engage in good water-saving practices while visiting the Muga river basin confirms the need for urgent action to raise awareness and improve water-saving habits. Over half of the tourists surveyed stated they would even be willing to pay extra if this money was invested in water-saving measures at the establishment, and almost three-quarters said that they would be willing to reduce their water consumption in return for a discount or other incentive. In our opinion, incentives encouraging guests to use water sparingly should be implemented by tourist accommodation establishments as they have proven to be very effective in some hotels, including international chain hotels, such as Expo Astoria hotel in Lisbon (a member of the Expogroup Company, Lisbon, Portugal) and numerous American hotels such as Starwood and Marriott [55]. In this second case, guests who reuse towels and bed linen or opt out of daily room cleaning receive a discount at the bar. Another strategy would be to reward guests with discounts on future stays, fulfilling thus two objectives: reduced water use and operating costs on the one hand and a greater likelihood of repeat and new bookings on the other.

Environmental awareness campaigns in the tourist accommodation industry should prioritize hotels, and signs or other information highlighting the problems of water shortages and encouraging careful water use in rooms should be mandatory. Hotels could also provide customers with information on the amount of water used at the end of their stay to encourage more efficient use. The installation of water sub-meters to monitor individual use could be used to offer discounts on stays or other services. Establishments should ensure that this information, alongside any other relevant programs, is clearly explained to guests on and before their arrival (e.g., on hotel websites and booking platforms).

Considering that tourists from outside the Mediterranean basin were the least likely to engage in water-saving practices and had the lowest levels of awareness, campaigns highlighting the need to use water sparingly while on holiday should target tourists in their countries of origin. Transit campaigns could also be effective. Leaflets explaining the problem of water shortages at the tourist's destination and stressing the need to use water carefully, for example, could be distributed on planes, trains, and ships. The Balearic Island Government launched a particularly interesting campaign at Palma de Mallorca Airport in the summer of 2019 that consisted of placing large transparent suitcases comparing water levels in the Balearic Islands and the tourists' country of origin on baggage reclaim belts as the tourists awaited their luggage [56]. The campaign caught the attention of many tourists, who, when interviewed afterwards, stated that they had been unaware of this problem and seemed agreeable to the idea of acting differently while on holiday to help safeguard the islands' water supplies. Campaigns of this type clearly lead to heightened awareness and ultimately encourage guests to use water more carefully when they arrive at their accommodation.

Finally, the results of this case study, together with findings from previous studies in the area, confirm the need to increase knowledge and awareness of the effects of climate change on water supply among both accommodation owners/managers and guests. They also highlight the importance of implementing mitigation and adaptation measures and ensuring that these efforts lead to a positive response by guests. For this to occur; however, coordinated action by all key stakeholders in the tourism industry, including owners, managers, guests, staff, and public bodies, is necessary.

Despite the above, it is likely that the negative economic consequences of the COVID-19 crisis will trigger a shift in priorities among hoteliers and other accommodation owners. It is important thus to continue to drive home the message of the importance of advancing towards sustainability and efficient water use and there is no reason steps in this direction cannot be included in general restructuring efforts. It is essential not to lose sight of the fact that solutions for combating major structural crises that affect the tourism industry and society as a whole are closely linked to efforts to combatting climate change and achieving a more sustainable use of natural resources.

8. Limitations

Our findings should be interpreted within the context of the time when our surveys were conducted (July to September, 2018). Since then, two major events have taken place. The first was an extratropical cyclone (Gloria) that wreaked havoc on the local economy and environment in January 2020 and the second is the COVID-19 pandemic, which has led to severe lockdown measures in Spain (starting on March 14, 2020), with a dramatic impact on the tourism industry. Future research should perhaps take account of these events to determine their potential influence on water-saving practices.

Although our sample was adequate for the general purpose of this study, our results may have been biased by overrepresentation of certain types of accommodation or reasons for stay. The number of participants staying at rural lodgings, for example, was lower than that of those staying at campsites or hotels, primarily because of the fewer guests and the difficulty of conducting on-the-spot interviews at rural properties. It should also be noted that our surveys were conducted the peak tourist season (June to September). Targeting tourists at other times of the year could reveal different profiles of guests. Our sample did not include 5-star hotels, as the two hotels of this category in the Muga river basin declined our invitation to participate in the study.

A final limitation is that our results may have been influenced by social desirability bias. Because we conducted on-site interviews, there may be some discrepancy between what the tourists said they do and what they actually do. In other words, this lack of anonymity may have prompted more "politically correct" answers, with respondents exaggerating good habits [57,58]. Nevertheless, one clear advantage of on-the-spot interviews is that they have higher response rates and provide more opportunities to clarify doubts and therefore obtain more information than self-administered questionnaires [59].

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Agricultural Water Allocation under Cyclical Scarcity: The Role of Priority Water Rights

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Abstract: Water is becoming an increasingly scarce resource worldwide, suggesting that water rationing methods should be revised to improve water allocation efficiency, especially during cyclical scarcity events (droughts). The proportional rule is the most widely used rationing method to allocate water in cases of water scarcity. However, this method fails to achieve Pareto-efficient allocation arrangements. Economic theory and international experience demonstrate that implementing security-differentiated water rights could improve allocative efficiency during cyclical scarcity periods. Moreover, it has been proven that this kind of priority rights regime is an efficient instrument to share risks related to water supply reliability, and can thus be considered as an adaptation measure to climate change. This evidence has enabled the development of an operational proposal for the implementation of security-differentiated water rights in the irrigation sector in Spain, as an alternative to the current rights based on the proportional rule. This proposal draws on the Australian case study, which is the most successful experience worldwide. Nevertheless, the insights obtained from the analysis performed and the proposal for reforming the water rights regime are applicable to any country with a mature water economy.

Keywords: water scarcity; water management; water rights; water supply reliability; irrigation agriculture; allocation rules; priority rights; Spain

1. Introduction

Water is becoming an increasingly scarce natural resource in many regions worldwide. The driving forces behind this are population growth and economic development, since both factors lead to a growing demand for water-intensive goods and services, most notably agro-food products (irrigation) [1]. As a result, we have witnessed over recent decades a marked increase in global water abstraction and consumption. Within this framework, supply-side measures (i.e., building new infrastructure like reservoirs and waterways to satisfy new human needs) are no longer a viable option in regions with mature water economies, where no further increases in resource availability are feasible either from an economic (prohibitive investment costs) or an environmental (conservation of water-related ecosystems) point of view. In these circumstances, river basins are said to be 'closed' [2], and new demands can only be met by reducing the existing ones through the implementation of so-called demand-side instruments, such as water pricing, water trade (water markets and water banks) or incentivizing water-saving technologies [3,4].

The closure of river basins has become common practice in the Mediterranean and semi-arid climate regions of developed countries such as Australia, Spain, or the United States (specifically western states). One thing all these territories have in common is competitive irrigated agriculture consuming up to 80% of total water use [5]. As such, there is strong competition for water between the irrigation sector and other users (urban consumption, other economic activities such as tourism, and the environment), evidencing the existence of 'structural' or 'permanent' water scarcity. Moreover, structural water scarcity is getting worse because of climate change. IPCC projections [6] for these regions indicate a decrease in precipitation and water availability, while the progressive temperature rise will increase irrigation water needs, resulting in greater demand for irrigation water.

In all these structurally water-scarce regions, water shortages are more severe during drought periods. In these episodes of 'cyclical' water scarcity, demand far exceeds water availability, and competition for the use of the resource becomes acute. Furthermore, according to climate change predictions [6], drought periods in these regions are expected to become more frequent and intense.

When water availability is lower than water demand, resources have to be rationed and allocated among users' needs. This is especially challenging during drought periods when the supply-demand gap or water deficit reaches its highest values [7].

Water is a complex economic good needed for economic activities as an input in many production processes (e.g., irrigation and industry). It also provides social and ecosystem services (e.g., drinking and sanitary water or ecological inflows). For this reason, water usually enjoys a distinctive legal status, managed under the public trust doctrine aimed at ensuring efficiency, equity, and environmental sustainability [8]. Under this doctrine, structural water scarcity is managed through water rights (or water entitlements) granted by a public authority responsible for allocating the average (or normal) water availability among socially recognized users, and preventing those who are not right holders from abstracting and using this resource. Current water rights regimes are often criticized because they are shaped by historical preferences and usage patterns that do not meet the needs of today's society, and they are poorly designed to cope with changing conditions such as new water demands or climate change [7]. All this justifies the need to reform water rights regimes, aiming at a more rational and sustainable allocation of scarce water resources in the long run [9,10].

Water rights regimes, in addition to determining who is allowed to use water resources, also establish how much water is available for each right holder in case of shortages (drought periods), when the total volume of water available is lower than the sum of the water volumes granted by the individual entitlements. Thus, water rights regimes also stipulate which of several existing rationing systems is to be implemented among right holders for scarcity management. Possible alternatives in this regard include proportional sharing or sequences of priority uses, sometimes combined with a water allocations trade. This paper is focused on analyzing these alternative policy options to cope with cyclical water scarcity, when the competition for water is at its most acute and rationing is most challenging [11].

Economics is "the science which studies human behavior as a relationship between ends and scarce means which have alternative uses" [12]. Thus, economics can play a key role in analyzing how scarce water resources ('scarce means') should be allocated among the demands from various users ('alternative uses') considering the desired policy objectives (the 'ends'). This justifies the application of economic theory to the analysis of alternative policy options for water rationing during drought periods.

Within this context, the objective of this paper is twofold. First, we outline a framework for the water allocation instruments and rules that can be implemented during drought periods to ration the scarce available water resources. For this purpose, we rely on economic theory, relating the different rationing alternatives to the policy objectives than can be achieved.

Second, the above-mentioned theoretical framework is used to analyze the water allocation and rationing system currently implemented in the Spanish irrigated agriculture sector. This allows us to explore how these instruments and rules could be improved to minimize drought-driven social welfare losses.

The choice of the case study analyzed here is justified for several reasons. First, because Spanish irrigated agriculture, like any other irrigated agricultural system in the Mediterranean region, is prone to be affected by frequent and intense drought episodes, with future projections indicating that this risk is likely to increase due to climate change [13]. Second, because of the importance of this sector in Spain as it covers more than 3.8 million hectares (17% of the national agricultural area), generating a total production valued at around 16 billion Euros annually and employing 415,714 workers [14]. Thus, any water supply gap affecting the irrigation sector entails relevant losses in terms of agricultural income and employment. And third, because there is consistent evidence showing that Spanish irrigators are willing to pay to reduce their exposure to droughts, taking into account the fact that there is no risk management instrument available (e.g., insurance) allowing them to cover potential losses related to water supply gaps [15].

2. Economic Foundation for Water Allocation Under Cyclical Scarcity

2.1. A Flowchart Framing Water Allocation Instruments and Rules

The main purpose of water policy is (or ought to be) to help ensure that water-related activities lead to a 'socially optimal outcome'. In this sense, it is widely acknowledged that in implementing this kind of policy, policymakers seek to successfully balance two conflicting objectives: economic efficiency and distributional equity. Thus, in order to promote a socially optimal outcome or welfare within this framework, policy design must achieve the appropriate trade-off between efficiency and equity objectives. This trade-off, based on society's concern for both objectives, is (or ought to be) expressed through the policy-makers' guidelines [16].

Establishing the appropriate trade-off between efficiency and equity is the core of normative economics, and it is the starting point of the positive economic analysis aimed at identifying the most suitable policy instruments for implementation in the real world to achieve the socially optimal outcome (i.e., social welfare maximization). Within this policy analysis framework, the water allocation instruments and rules under cyclical scarcity can be framed as shown in Figure 1.

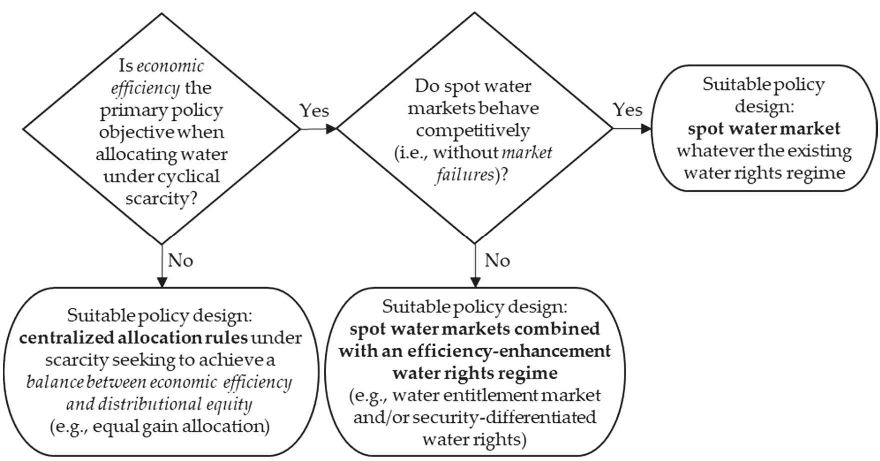

Figure 1. Flowchart framing suitable water allocation instruments and rules under cyclical scarcity.

In sum, the flowchart in Figure 1 shows that if economic efficiency is the primary policy objective when allocating water under cyclical scarcity and competitive spot water markets can be developed in a real setting, this trade-based allocation instrument yields the optimal policy outcome. Under these circumstances, the market can reallocate scarce water resources among users irrespective of their initial allocation (water rights regime), with the final allocation enabling the maximum aggregate net benefit from water use (i.e., economic efficiency) [17]. However, if there are market failures (relevant externalities, high transaction costs, or barriers to trade), spot water markets could fail to achieve economic efficiency, with the final outcome depending on the initial allocation of water resources available. Thus, the question of how water rights regimes are defined (i.e., how water is initially allocated) becomes a key issue [18]. In the presence of market failures, spot water markets should be combined with water rights regimes that minimize those failures, enabling the final allocation of scarce resources to enhance economic efficiency (higher aggregate net benefit from water use).

Moreover, policymakers can also consider equity as another relevant policy objective to be achieved when allocating water during drought periods. In this case, policy action based purely on market instruments is not recommended, since trading instruments on their own usually lead to an inequitable final allocation of water, exacerbating income gaps between regions and economic sectors [19]. In light of this circumstance, the public trust doctrine can be justified within the water sector, with the water rights regime being regulated to allow a public authority to allocate scarce water resources based on public interest criteria and ban any possibility of water trade (i.e., the initial water allocation remains unchanged). In this sense, there are several allocation rules with different characteristic properties in terms of efficiency and equity that can be implemented. Depending on the policy guidelines regarding the trade-off between economic efficiency and equity objectives, the most suitable allocation rule can be chosen as the policy option to ration water among right holders under scarcity conditions.

The following subsections provide more detailed explanations justifying the suitability of each policy option suggested.

2.2. Spot Water Markets as Allocation Instruments

The market is an economic institution widely used to allocate economic (scarce) goods, including natural resources, among their alternative uses based on a decentralized price mechanism. The widespread use of market instruments is supported by economic theory, more specifically through the First Theorem of Welfare Economics, which states that if there are markets for all commodities and all these markets are competitive, then the equilibrium of the economy is efficient [20]. This theorem explains why water markets are advocated as efficient instruments for water allocation under scarcity settings.

In this sense, it is worth clarifying that the type of efficiency potentially achieved by spot or allocation water markets is Pareto efficiency, also referred to as allocative efficiency. This means that a property of any of the various resource allocation arrangements that could be achieved through these water markets is that there is no other feasible allocation which would make some individuals better off and no individuals worse off. Achieving this kind of efficient arrangement is possible because markets create a system of economic incentives to allocate water to higher value uses through mutually advantageous trade operations for sellers and buyers, at least until the equilibrium price is reached and further gains from trade are exhausted. Within this decentralized allocation framework, the marginal values of all water users became equal to the equilibrium price, maximizing their net benefits, and thus the aggregate net benefit from the use of the water available [21].

Moreover, Pareto-efficient market solutions have two interesting features that are worth pointing out. First, the final allocation arrangements achieved through trade are independent of the initial allocation of resources (i.e., the distribution initially set by water rights) [17]. Second, the equilibrium prices reached are dynamic, always reflecting the full opportunity cost of water (i.e., the scarcity rent). This makes spot water markets flexible economic instruments, which allow a timely, decentralized adaptive management approach for every local situation.

Given all of the above-mentioned characteristics, the economic literature has identified competitive water markets as the most efficient water allocation instruments to cope with water shortage situations (drought periods) [22–24]. They are considered especially suitable for implementation in cases where there is no relevant concern about equity-related objectives or, simply, these objectives are pursued through other, horizontal policies such as taxation and welfare programs targeted at improving social equity. In fact, substantial economic efficiency gains from water trade have been acknowledged in empirical analyses from around the world (e.g., [25–27]), including those specifically focused on the irrigation sector (e.g., [28–31]).

In any case, the consideration of spot water markets as efficient allocation instruments during scarcity periods needs to be further examined, taking into account a number of key issues. First, the complex nature of water resources, which creates numerous sources of market failures, meaning that the actual spot water markets allocate resources inefficiently. In this regard, there are two predominant sources of market failure, common to all water markets worldwide [21,32]:

1. Water is used for a wide variety of public purposes (e.g., instream flows for maintaining ecosystems or recreational activities). These water uses are public goods that cannot be exchanged in markets, and thus they have no market price signaling their relative scarcity. This means that the marginal benefits from these uses are understated when allocating water through the markets, leading to the underproduction of these public goods.
2. Market reallocation of water resources usually generates externalities or third-party effects (e.g., changes in the quantity and quality of return flows or economic side-effects in regions selling water). As the level of externalities generated is not controlled by any market equilibrium price, it cannot be concluded that the allocation solutions achieved through trade are efficient.

In the presence of either of these two failures, stand-alone spot water markets are neither efficient nor socially acceptable instruments for managing water resources under shortage scenarios [32].

Second, it also worth pointing out that water market operations involve transaction costs: the costs over and above the market water price that the water buyers must bear when purchasing water allocations, due to water conveyance costs, search and information costs, bargaining and decision costs, and enforcement costs, including contracting [33,34]. Depending on the institutional arrangement and the elasticity of supply and demand, these transaction costs can also be borne by water sellers, which would negatively affect their revenues from sales. The existence of transaction costs is relevant for market activity since they involve an inward shift in the demand curve and an outward shift in the supply curve. This results in a reduction of the market activity since the only transfers that take place are those where the differences in marginal values (i.e., potential gains) exceed transaction costs. Therefore, the higher the transaction costs, the thinner the market and the lower the net gains from reallocation [35]. The existence of transaction costs thus affects the final allocations of water, making them more dependent on the initial assignment of resources (i.e., the distribution of water rights), thereby limiting market efficiency [18].

To cope with widespread market failures and high transactions costs, different strategies have been suggested to minimize their efficiency-limiting effects: command-and-control regulations (e.g., setting minimum ecological instream flows), economic incentives (e.g., taxing polluting activities or subsidizing water-saving technologies), or public sector participation in water markets, translating the social values of water into market values. In this paper, however, we focus on how to combine spot water markets with alternative water rights regimes to improve the economic efficiency of water use under cyclical scarcity situations. To this end, we rely on the works of Freebairn and Quiggin [36] and Lefebvre et al. [37], who studied the effects of the implementation of water rights with different levels of supply security as a complement to water markets, showing that these kinds of water rights reduce the effects of market imperfections compared with proportional water rights. This evidence supports the need for the application of water rights regimes based on priority allocation rules as a way of improving economic efficiency under water shortage scenarios.

The third and last key issue regarding the efficiency of spot water markets during scarcity periods is related to society's equity concerns. In this sense, it is worth noting that efficient markets do not necessarily lead to socially optimal water allocation or Pareto-optimality, the best performing allocation arrangement in terms of the social welfare function, where equity concerns are also considered [20]. In fact, the Pareto-efficient allocations achieved through a competitive spot water market may be highly inequitable, meaning that these market solutions may not maximize welfare functions based on value judgments which prioritize equity [38]. In cases where the allocation arrangements achieved through market transactions are socially perceived as 'unfair', state intervention is also justified. One policy option is to maintain the market as the water allocation instrument and amend its efficient outcome in welfare terms using income redistribution programs (e.g., through the tax system and welfare state instruments). The other option, as displayed in Figure 1, is to discard the market as an allocation mechanism and replace it with centralized allocation rules (i.e., water rights regime) that are aimed at ensuring the highest possible social welfare associated with the use of water, as explained in the next section.

2.3. Centralized Allocations Rules

When policymakers consider not just efficiency but equity too as a relevant policy objective, the revision of centralized allocation rules emerges as an interesting alternative to water markets for improving resource allocation in drought situations. These rules are exogenous regulatory mechanisms through which regulators can alter how water use rights are shared, seeking to achieve the greatest possible social welfare (Pareto-optimality) associated with the use of water [39].

The problem of how to fairly allocate available resources in a system that cannot satisfy all the demands or claims of the beneficiaries is a classic question, which has been widely analyzed in the economic literature as the 'bankruptcy problem'. The original framing of the problem relates to a situation in which several agents claim different amounts of money that together exceed the liquidation value of a bankrupt company, and this liquidation value must be divided among the agents. However, bankruptcy-like problems can be found in many other real-life problems, where the application of this approach has proven suitable. The challenge of water rationing under shortage conditions, seeking a fair allocation of the total water deficit (the difference between the total demand and the available resource) among water rights holders, is one such field of application [40], as has been shown in various empirical studies (e.g., [41–44]).

Within the rationing methods proposed to solve the bankruptcy problem, we can distinguish between symmetric and asymmetric ones. Symmetric methods are those that are based on the 'equal treatment of equals' axiom, which promotes equal shares to equal demands. On the other hand, asymmetric methods are those that do not comply with this axiom, and thus shares are allocated on the basis of priority criteria, whether in relative or absolute terms.

2.3.1. Symmetric Methods

Within the symmetric distribution methods, there are three predominant ones. These are the proportional, equal gains, and equal losses methods [45,46]. The proportional (P) method is the best known and it is based on all claimants being assigned an amount (water allocation in our case) proportional to their claim (water rights). In the equal gains method (EG, also called 'uniform gains' or 'constrained equal awards'), all claimants receive the same amount, as long as it does not exceed what is claimed. Similarly, in the equal losses method (EL, also called 'constrained equal losses') all claims are trimmed by the same amount, on the condition that no-one should receive a negative amount.

These three allocation rules (P, EG, and EL) comply with four basic properties or axioms that make them suitable for implementation for agricultural water sharing [47,48]:

1. Consistency, when the rationing method allocates the same volume of water irrespective of whether it is applied to all claimant irrigators at the same time or separately for different subsets of irrigators.
2. Independence of scale, which can be interpreted as independence regarding the unit of measure of the resource rationed (e.g., cubic meters, megaliters, acre-feet, etc.). This implies that any proportional increase in water availability and demands results in the same proportional increase in water allocations.

3. Composition down (or 'upper composition') is an invariance property regarding changes in the availability of the resource being distributed. This occurs when the individual water allocations are calculated in advance based on the expected water availability, but in the end, there is actually less water to share than initially assumed. In these situations, reapplying a rationing method that complies with this axiom yields the same final allocation arrangement as would have been achieved in a single step if it had initially been implemented based on the actual volume of water available.
4. Composition up (or 'lower composition'). This is similar to the previous one, but this axiom applies when the volume of water initially allocated is lower than the volume of the resource eventually available for sharing. Under these circumstances, the allocation of the additional volume of water available by reapplying a rationing method meeting this property results in the same final allocation arrangement as would have been achieved in the beginning if the initial calculation had been based on the actual amount of water available.

In the literature regarding the allocation of agricultural water, there are a number of empirical studies that analyze the efficiency of symmetric rules. Goetz et al. [49,50] and Martínez and Esteban [51], through applications implemented in three different irrigation districts in Spain, demonstrate that the EG method is more efficient from an economic perspective than the proportional one. However, in both cases, the two symmetric allocation rules considered substantially reduce the economic efficiency compared with market allocation. Similarly, Alarcón et al. [52] compare the proportional and the EL methods with the optimal allocation where total economic loss is minimized (mimicking the results from a competitive spot water market) in another Spanish irrigation district. They conclude that the implementation of the proportional method results in large efficiency losses compared to market allocation. While the market allocation also outperforms the EL method in efficiency terms, the efficiency losses are less than with the proportional method.

All these empirical works also prove that the more heterogeneous the irrigators and the scarcer the water resources, the larger the efficiency losses of the symmetric methods compared to market allocation.

Finally, it is worth citing the work by Madani and Dinar [39], who compare the performance of the proportional and EG methods for groundwater management using a stylized numerical example. Their results differ from previous evidence, showing that the proportional method outperforms the EG method in terms of economic efficiency and equity criteria.

2.3.2. Asymmetric Methods

If the axiom of equal treatment of equals is not met, we are dealing with asymmetric rationing methods. In these cases, agents are classified into priority classes according to exogenous criteria, with their demand being met lexicographically following a priority order also set exogenously. That is, the demands of the agents with the highest priority are met first and, once fully satisfied, the remaining resource is allocated to the following agents according to a decreasing priority order criterion [48].

The most asymmetric rule is the full sequential allocation, where every single agent is considered as a different class [53]. This is the theoretical foundation of the prior appropriation doctrine used in the Western United States to define water rights. This doctrine is based on the legal principle expressed by the Latin phrase "*qui prior est in tempore, potior est in jure*", which means "he/she who is first in time is first in right". Following this doctrine, water right holders in the Western United States are ordered along a line according to the seniority of their rights; the longer the right has existed, the higher the priority assigned to it.

Priority rules can also be established considering a reduced number of priority classes (two or three, for instance, depending on the type of users: urban, environment, and economic activities). In this case, agents in different classes are treated differently according to the priority order criterion, but agents classified in the same priority class are treated under the axiom of equal treatment of equals, using any of the above-mentioned symmetric methods (P, EG, or EL methods).

Other asymmetric methods include those based on weights indicating the 'relative priority' (as opposed to the 'absolute priorities' outlined in the preceding paragraphs) that should be given to agents [47]. In all cases, each claim is multiplied by the exogenous weights assigned to the agent holding it, and shares are calculated following any rationing method, with the condition that no-one should receive more than his/her claim. Examples of methods that involve this procedure include the weighted proportional method, the weighted gains method, or the weighted losses method.

All these asymmetric methods also meet the four above-mentioned desirable properties for water management; namely, consistency, independence of scale, composition up, and composition down [48].

Calatrava and Garrido [54] provide an example of the implementation of the weighted proportional method within an irrigation district, where ligneous crops are given relative priority over horticultural crops, and the latter over extensive annual crops. These authors demonstrate that this allocation method achieves greater economic efficiency than the proportional one, but lower than market allocation.

2.3.3. Rationing Methods and Social Choice

The rationing problem has also been analyzed from the perspective of social choice theory, analyzing the role of asymmetric information in the implementation of allocation rules and the contribution to Pareto-efficient arrangements [55].

In bankruptcy problems, the allocation rules are unequivocally applied to the specific and public demands of each claimant. However, in many other rationing problems, as is the case of water sharing, the demands of the agents involved in the allocation are characterized by asymmetric information since only the claimants know their real needs (the optimal quantities they want to demand). In this context, the rationing rules must comply with the property of strategyproofness, which denotes that agents have a single preference that dominates over all other strategies and that these agents have no incentive to claim more or less than they really need. Of all the symmetric rationing methods discussed above, only the EG method is strategyproof, in addition to complying with the properties of Pareto-efficiency [56]. Likewise, all the above-mentioned asymmetric allocation rules are strategyproof, but none of them are shown to be Pareto-efficient. However, Barberà et al. [57] have developed a sequential rule that fulfills the properties of both strategyproofness and Pareto-efficiency. This asymmetric rule is similar to the EG, except for the fact that the agents, in addition to having different preferences on the quantity of the resource demanded, also have rights over different maximum allotments.

Goetz et al. [50,58] apply the sequential rule proposed by Barberà et al. [57] in two irrigation districts in Spain and compare it with the P and EG methods, providing evidence that this rule is more efficient than the two symmetric ones, especially if there is substantial heterogeneity among irrigators. Nevertheless, they conclude that the allocation obtained by implementing this sequential rule is less efficient than market allocation.

3. Agricultural Water Management in Spain

In line with the public trust doctrine, the Spanish Water Law (Royal Legislative Decree 1/2001) establishes that all water resources are considered to lay in the public domain. Thus, the use of water for economic activities requires an administrative concession or water right. These water rights are granted by basin authorities according to the river basin management plans (RBMP), taking into account the rational exploitation of resources (i.e., in relation to average water availability based on current infrastructure like reservoirs and waterways), however, the water rights held do not guarantee the actual availability. Logically, the effective water use by right holders is subject to the actual availability of the resource (i.e., water stored in reservoirs). When there is a water shortage due to hydrological drought events (i.e., below-average levels of water stored), the basin authorities temporarily limit the use of the water legally granted in the water rights, applying a combination of two rationing rules. First, right holders are classified into priority classes depending on the type of water use. Based on general interest criteria, the Spanish Water Law considers urban use (human consumption and industries connected to urban supply networks) to be the first priority, followed successively by agricultural uses (irrigation), electric power production, industrial uses, aquaculture, recreational uses, navigation and, finally, other uses not included the aforementioned categories. Therefore, under scarcity conditions (droughts), water allocation in Spain is managed by implementing a priority rule differentiating between types of use. Second, within each priority class, all right holders are rationed using the proportional method (i.e., when the total volume of water available for the class in question, once the demands of higher priority classes have been fully met, is not enough to meet the demands of the right holders).

Table 1 shows the water rights legally granted in the main Spanish river basins, with these rights divided into priority classes: urban, agricultural, and other uses. In this sense, it is worth highlighting the relevance of agricultural use, which accounts for 78.8% of total water rights at the country level. Furthermore, it can be observed that the river basins with the largest water allocations (Ebro, Duero, and Guadalquivir) are precisely those where irrigation water consumption represents the greatest share of the total, accounting for around 90% of the total water use in these territories.

Table 1. Water rights granted in the main Spanish river basins [1].

River Basin	Urban		Agricultural		Other Uses		Total
	hm^3/year	%/total	hm^3/year	%/total	hm^3/year	%/total	hm^3/year
Ebro	614	(7.3%)	7679	(91.7%)	85	(1.0%)	8378
Guadalquivir	400	(10.6%)	3328	(88.2%)	43	(1.2%)	3771
Duero	285	(7.6%)	3426	(91.2%)	46	(1.2%)	3756
Tajo	994	(33.1%)	1912	(63.7%)	96	(3.2%)	3002
Júcar	572	(20.5%)	2182	(78.2%)	35	(1.3%)	2789
Guadiana	254	(10.8%)	2022	(85.7%)	82	(3.5%)	2359
Segura	238	(14.9%)	1353	(84.6%)	9	(0.6%)	1600
Andalusian Mediterranean	279	(25.3%)	770	(70.0%)	51	(4.6%)	1100
Other river basins	1948	(48.2%)	1595	(39.5%)	500	(12.4%)	4042
Total Spain	5584	(18.1%)	24,266	(78.8%)	948	(3.1%)	30,797

Source: Dirección General del Agua and Centro de Estudios Hidrográficos [59]. [1] 1 hm^3 equals 1 Mm^3, or 1 GL, or 810.71 acre-feet.

In Spain, the concession of new water rights by the basin authorities is only possible if two conditions are met: (i) the new water uses contribute to the general interest criteria set in the RBMP, and (ii) the new water demand can be satisfied in accordance with the reliability criteria set at country level, taking into account the actual availability of water resources in each basin (i.e., based on climate, geography, and available infrastructure for water storage and transport). This second requirement is a constraint on many Spanish basins that have been officially declared 'closed' since there are no further possibilities of increasing the water supply.

Considering the requirements regarding supply reliability, all water rights granted can be fully satisfied in 'normal' (i.e., close to average) hydrological years. In fact, it is only in cases of prolonged drought episodes that there are problems meeting all these demands, making it necessary to ration water allocations for some users, starting with the lowest priority uses. In this regard, taking into account the legally established priority of urban over agricultural uses, in river basins where a major share of water rights is assigned to agricultural use, the supply of water for urban use is practically assured even in the most extreme drought scenarios. Thus, in situations of cyclical scarcity caused by hydrological droughts, water supply restrictions almost exclusively affect allocations for agricultural purposes.

In accordance with the European Water Framework Directive, Spanish law also establishes that basin authorities must approve drought management plans (DMP) as a complement to their RBMP. These plans specify the way in which water resources must be managed and allocated during periods of scarcity. For this purpose, a set of drought indicators have been defined to provide information about the current scarcity scenario [60]: normality (absence of scarcity), pre-alert (moderate scarcity), Alert (severe scarcity), and emergency (extreme scarcity). If the indicators point to any scenario other than Normality, the basin authority must enact the drought management measures set out in the DMP to minimize the environmental, economic, and social impacts of scarcity.

In the event of alert or emergency scenarios, the basin authority reduces water allocations for irrigation, with all agricultural right holders receiving equal rations determined using the proportional method as water allocation rule. Thus, allocations to all irrigators within the same water use system (management units within river basins) are proportionally reduced to maintain the reserves needed to meet higher priority uses (i.e., urban use). The implementation of this proportional rule, however, does not produce an economically efficient distribution of the water available for the agricultural sector since the irrigators being allocated the rations are quite heterogeneous. This heterogeneity is mainly due to the varying pedoclimatic conditions found within a single water use system (e.g., the *Regulación General* water use system in the Guadalquivir basin comprises 723,951 irrigated hectares), although differences in farm size and farmers' psychological characteristics (e.g., risk aversion) may also have an influence. In these circumstances, proportional water rationing leads to quite different impacts on farmers depending on their water productivity. In fact, the losses caused by water supply cuts differ notably between extensive and intensive agriculture: farmers dedicated to the extensive production of herbaceous crops (cereals, industrial crops, etc., with low marginal productivity of water) face moderate losses, while the losses for farmers producing intensive crops (vegetables or fruits, with high marginal productivity of water) are potentially very high. This explains why this allocation method is not optimal from an economic point of view since it fails to minimize the losses derived from water scarcity.

In an attempt to partially solve this inefficient allocation during cyclical scarcity periods, spot water markets and public water banks were made legal in Spain in 1999. However, their performance as a water reallocation instrument has been rather disappointing [61]. In fact, water trading has been active only during drought periods, and even under these severe scarcity situations, market activity accounted for less than 1.0% of total water use. The most intense trading occurred during 2007, an extremely dry year, when water exchanges accounted for 248 Mm^3 (0.78% of total water use in Spain for this year, although the share of water traded reached 4% for some basins in southeastern Spain), with the water price in these two market instruments ranging from 0.12 to 0.27 Euros/m^3 [62,63]. These figures show the narrowness of the water markets in Spain, suggesting that transaction costs and multiple barriers to trade are hampering their effective functioning.

Moreover, following the 2019 general election in Spain, a new left-wing coalition government was formed. This new government is founded on a coalition agreement document signed by the parties sharing the political power, which sets the policy guidelines for the current legislative term. This agreement establishes the political intention to ban water markets on the basis that water "should not be considered a commercial asset". Thus, a reform of the Spanish Water Act is expected, forbidding water trade among water users (some doubts remain regarding the public water banks operating in Spain during droughts). This legislative reform has not been accomplished yet, and the policy agenda in the short-term has changed because of issues related to the Covid-19 pandemic. In any case, the Spanish government still intends to launch this water policy reform and approve it before the end of the current legislative term in 2022. This expected legal change triggers the need for new alternative designs of water rights regimes to prevent the efficiency losses caused by the implementation of the proportional rule during drought events.

4. Alternative Water Allocation Methods: The International Experience

The Western United States and Australia provide valuable examples regarding agricultural water management. Both countries share some key characteristics with Spain (semi-arid climates, a mature water economy, a large and competitive irrigated agriculture sector, and severe cyclical water scarcity problems) [5]. As such, they can be considered as suitable benchmarks to learn about allocating agricultural water during drought periods. This section briefly describes the asymmetric allocation rules implemented in each of these countries, before critically analyzing their pros and cons, as well as their suitability for potential implementation in Spain.

4.1. Western United States

In the western United States, water rights are mainly governed by the 'prior appropriation doctrine' (PAD), which establishes a fully sequential allocation method. The priority is thus determined by the chronological order in which the rights were granted, from the most senior (the longest-standing rights have the highest priority) to the most junior (the most recent have the lowest priority).

In addition to appropriative water rights, there are two other minority types of water rights in the western United States: 'pueblo' rights and 'federal reserved' rights. The first are water rights initially granted to the Spanish and Mexican 'pueblos' (settlements), and later legally recognized to preserve the traditional water rights held by some cities (e.g., Los Angeles) and native American communities (e.g., New Mexico Pueblos) [64,65]. The federal reserved rights are established when the U.S. federal government reserves public land for uses such as Indian reservations, military reservations, or national parks, with each reservation being granted the water rights needed to satisfy the purposes for which it is created [66]. Moreover, riparian water rights are also used in the states on the West Coast (California, Oregon, and Washington) and more humid parts of the Dakotas, Nebraska, Kansas, Oklahoma, and Texas.

The origin of the PAD dates back to the settlement of the American West, where it was used as a simple and efficient way to allocate water consistent with the Ricardian theory of land rent. Indeed, at the time, this doctrine contributed to an efficient allocation of water resources, since the first lands to be irrigated (thus holding more senior rights) were likely be the most productive and profitable ones. However, more than a century later, the most profitable uses of water are not necessarily those that hold the most senior water rights. For this reason, water allocations based on prior appropriation today have drawbacks from the point of view of economic efficiency [67]. Moreover, PAD-based water rights generate heterogeneity in risk-sharing among water users (the more junior right holders run a higher risk of receiving insufficient allocations), which may also contribute to an economically inefficient allocation of water [68].

To minimize the aforementioned inefficiency problems, water rights in the western United States are not tied to the land, with existing spot and permanent water markets allowing allotments and rights transactions, respectively. However, in most of these states, there are institutional barriers to the transfers of water rights in order to prevent third-party effects, which limit the allocative role of the market and, thus, economic efficiency [69].

Water rights in the western United States are also governed by the continuous beneficial use doctrine ('use it or lose it'), which establishes that these rights remain in force only as long as the beneficial use continues. Nevertheless, the application of this doctrine does not always contribute to an efficient allocation of water, given that it may encourage excessive water consumption aimed at maintaining the right. Furthermore, it can also lead to inefficiency if right holders intensively use water for uses legally considered 'beneficial', but with low marginal value [70].

4.2. Australia

The most noteworthy type of water rights in Australia are the security-differentiated priority rights which have been in place in the state of Victoria since 1994, and New South Wales (NSW) since 2000. In the rest of Australia, agricultural water is allocated based on the proportional rule.

In these two southern Australian states, agricultural water allocation follows a two-step procedure. First, rights are ordered according to their priority level. Second, the amount of water available for each priority class is shared out through proportional rationing. The implementation of this allocation mechanism involves two types of water rights for agricultural uses with different levels of security (i.e., reliability): high-priority and low-priority rights, although they have different names according to the state [71]. For instance, in NSW, high-priority water rights (officially, 'high-security access licenses') account for 9% of total rights granted for agricultural uses, with the remaining 91% being low-priority rights or 'general-security access licenses'. The estimated reliability of agricultural high-security access licenses in NSW is 95–97%, meaning that farmers can expect to receive their full allocations at least 95 years out of 100. On the other hand, the average reliability of general-security access licenses is around 70% [71,72].

The water allocation procedure starts at the beginning of the season, when high-priority right holders are allocated 100% (95% in NSW) of the nominal quantity established, while those with low-priority rights are assigned only a small percentage of the quantity established in their rights. Over the course of the irrigation season, the water allocated to the latter type of rights is increased depending on the actual water availability, following a proportional rule. In the event that there is not enough water available at the beginning of the year to provide full allocations to high-priority rights (in extreme drought situations), the available water is shared proportionally among the high-priority rights but no water is allocated to low-priority ones [73].

It is also worth commenting that once the water has been allocated, there are no restrictions on reallocating the resource through the existing spot water market, which allows water transfers from low to high value uses between any type of right holder and across states [74]. Moreover, in Australia there is a permanent or water entitlement market that facilitates the transition towards different farming systems (e.g., change in farm size or crop mix) or just makes it easier to leave the farm sector [74]. Both markets have become quite active, with significant improvements in water property rights, trading rules, and market information, as well as reduced transaction costs over time. As a result, in an average year, around 30% of the announced water allocations and 10% of water entitlements are traded, facilitating economic efficiency in the short and the long run [75].

4.3. Pros and Cons of the Priority Allocation Methods

Water rationing methods based on priority levels have a series of pros and cons which merit analysis before their implementation in a real-world setting.

The main advantages are explained by Freebairn and Quiggin [36] and Lefebvre et al. [37], who argue that a water rights regime with different levels of priority is an interesting alternative for the allocation of water resources within the agricultural sector since it enables more efficient risk-sharing. This allocation mechanism can be used to offer irrigators a portfolio of different water rights establishing different priority levels, suited to their particular circumstances (vulnerability to water supply gaps and risk aversion). For instance, those irrigators running intensive high-value crops could reduce the risk related to water reliability by obtaining high-priority rights, with this risk then being transferred to lower priority right holders, who are better positioned to assume this risk (e.g., farmers with extensive annual crops). This mechanism to transfer water supply risk is much easier to implement than other risk transfer instruments, such as hydrological drought insurance or water options markets.

Moreover, this water rights regime reduces the number of transactions needed in the spot water markets, with the consequent reduction in transaction costs. Thus, the efficient risk-sharing arrangement generated by this regime also results in improved economic efficiency. This is especially relevant in countries where spot water markets have high transaction costs but low volumes of trade activity.

Finally, it is also worth noting that in addition to the above-mentioned short-term advantages, it has been observed that security-differentiated water rights offer a series of long-term benefits. The improvement in the reliability of supply for higher priority right holders enables them to invest in irrigation infrastructure and to transition towards higher added-value farming systems [76,77]. In this sense, it should also be pointed out that high-priority rights can facilitate investments since they act as capital assets that can be held as collateral to secure bank loans [37].

By contrast, it is important to highlight the potential drawbacks of a water rights regime with different levels of priority. First, the configuration of efficient portfolios of water rights requires the implementation of a flexible mechanism allowing users to modify the mix of different priority rights they hold (e.g., permanent or water rights market). These mechanisms usually involve significant transaction costs, limiting the efficiency improvements that can be achieved by any security-differentiated allocation method [78].

In this sense, it has been suggested that the most suitable design for priority-differentiated rights is the one based on two priority classes (as in Victoria and NSW), since by combining two types of priority rights water users can achieve any desired level of reliability, while minimizing the transaction costs related to the dynamic adaptation to the right mixes [79].

Likewise, it must be noted that the counterpart to the improvement in the supply reliability obtained by higher priority right holders is the loss of security for the rest of the water users. Thus, if there is no agreement as to some type of compensation from 'reliability winners' to 'reliability losers', the introduction of security-differentiated water rights could be politically and socially controversial.

5. A Proposal for an Alternative Allocation Method in the Spanish Irrigation Sector

The above analysis supports the reform of the agricultural water rights regime in Spain, with a shift away from the proportional allocation rule currently in force towards a priority rule allowing the implementation of security-differentiated water rights. Next, an operational proposal is introduced, specifying the characteristics that, a priori, can be considered more suitable for the Spanish case:

- Types of security-differentiated water rights and rationing rules for agricultural use: Two priority classes are proposed, distinguishing between high-security or 'priority rights' and low-security or 'ordinary water rights'. The rationing method applicable during scarcity periods would be similar to the one implemented in Australia, based on a combination of priority allocation between priority classes and proportional allocation within each class.
- Assignment of priority rights: The priority rights would be granted by the basin authorities, who would be responsible for ensuring the water supply to the right holders according to the RBMP approved.
- Initial distribution of priority rights: Considering the fact that most of the Spanish basins are closed, the proposed reform cannot increase the total amount of water granted through water rights. Therefore, it is proposed that all existing water rights should automatically be converted into ordinary ones, and that only a certain share of these existing rights should be allowed to be upgraded into priority rights through an auction procedure. In this sense, we suggest that only 10% of current ordinary rights in each water use system should be upgraded into priority rights, in order not to have an excessively adverse effect on the reliability of the remaining ordinary ones. The assignment of priority rights could be carried out through a uniform-price sealed-bid auction [80] considering the right holders' willingness to pay measured as a surcharge on the annual regulation tariff (*canon de regulación*) currently paid to the river basin authorities to finance the water storing and transport services provided by these public agencies. This additional public income would contribute to improving the public supply services provided, especially those related to the reliability of ordinary rights.
- Duration of the upgrade into priority rights: In line with the current legal framework in Spain (whereby water resources are publicly owned and water rights are granted by a public authority for a limited time, with a formal renewal subsequently required to continue using the water), it is proposed that those who win the bid in the auction procedure can hold priority rights for the next 20 years. This is considered a reasonable length of time to enable long-term investment planning in the agricultural sector (fruit orchards, irrigation technology, or specific agricultural machinery). If the water rights were legally renewed before the end of this term, the length of the priority rights would be subject to the renewal of the water rights (i.e., in no case would the assignment of priority rights imply the tacit renewal of concessional rights). In this way, the public assignment of water rights based on public interest would be preserved.

- Dynamics of the priority rights: After the initial assignment of priority rights, priority right holders would have the possibility of renouncing those rights (i.e., downgrading their priority rights into ordinary rights and stopping paying the surcharge on the annual regulation tariff). This downgrade should be done during the last month of any hydrological year (September). In this way, a reserve of priority rights could be established in each water use system, and the available priority rights could be auctioned again during the first month of the hydrological year (October). This alternative to the water right market is chosen for two reasons: i) it is expected to reduce transaction costs (addressing efficiency concerns), and ii) to preserve public interest (equity concerns).

In order to further clarify the proposal developed, Table 2 shows the main features of the current and the proposed water rights regimes for the Spanish irrigation sector.

Table 2. Current vs. proposed water rights regimes for the Spanish irrigation sector.

Feature	Current Water Rights Regime	Proposed Water Rights Regime
Types of water right	Ordinary	Priority and ordinary
Allocation rule	Proportional	Combination of priority allocation between types of water rights and proportional allocation within each type
Distribution of water rights	Granted at the discretion of the basin authority, based on public interest criteria	Priority right can be obtained only by current (ordinary) water right holder through uniform-price sealed-bid auctions
Duration of water right	75 years	75 years for ordinary rights and 20 years for priority rights
Dynamics	Virtually automatic renovation of water rights for new 75-year periods	After the 20-year period, priority rights became ordinary rights again. Possibility of renouncing priority rights and subsequent auction for allocation to other right holders
Fees	Annual regulation tariff for water storing and transport services provided by the basin authority	Surcharge on the annual regulation tariff for priority water rights and a reduction in the tariff to be paid by ordinary right holders

Another suggestion worth considering is that the new water rights regime proposed should allow individual rights to be defined as a combination (portfolio) of ordinary and priority rights (e.g., with 30% of the rights being priority and the remaining 70% ordinary). This would make it easier to engage collective users, such as the irrigators' associations (*comunidades de regantes*) that hold a single concessional right to supply water to a large number of farmers. Thus, by defining the water rights as portfolios, irrigators' associations could internally implement differential allocation rules according to the varying preferences of their members.

Finally, it should be noted that the above-mentioned proposal, which aims to strike a balance between the two main objectives of water policy (efficiency and equity), has the advantage of being compatible with current Spanish water legislation. In fact, the changes needed to implement the proposed reform could be easily done by updating the RBMP and DMP, where priority uses and allocation rules are defined for each basin.

Furthermore, another notable feature of this new water rights regime is that it acts as a complement to the spot water market already operating in Spain as instruments improving economic efficiency. If water markets were to be banned, as proposed by the new government, the proposal of security-differentiated water rights could be even more appealing since this instrument would be considered as a substitute mechanism providing water right holders the flexibility to adapt to the market and climate change.

The only drawback of the proposal is that the introduction of the priority rights would lead to a deterioration in the security of ordinary rights, which would provoke opposition from affected right holders if they are not properly compensated. In order to minimize this problem, it is suggested that the additional income generated through the surcharge on the annual regulation tariff to be paid by priority right holders should go towards improving the supply reliability of affected right holders, financing negotiated infrastructure, and funding management mechanisms.

Finally, it is also worth noting that the auction procedure suggested for upgrading into priority rights can be considered as a partially-market-based allocation of water rights, and thus could be contested by those who criticize the use of economic instruments as an alternative to the public sector action in the allocation and management of water resources [81,82]. This a sensitive topic in Spain where there is a strong social and political debate about the implementation of economic instruments to improve economic efficiency and allocative equity in water use, since water is a resource laying in the public domain (i.e., public-owned). In fact, some social and political actors see the implementation of these instruments as part of a wider project of 'privatization' and 'commodification' of water that favor the interests of some concrete agents, instead of the public interest they supposedly promote [83,84]. This circumstance calls for further discussion of the proposal introduced aiming at reaching the political consensus needed for the success of this policy reform. In this sense, it is suggested that the auction procedure considers some legal constraints leading to a socially optimal outcome. Thus, this allocation procedure makes it possible to contribute positively to the achievement of public interest objectives taking into account interregional and intersectoral related issues [85].

6. Conclusions

Water is getting scarcer in most of the Mediterranean and semi-arid regions around the world, since the demand for this resource is growing while its availability is declining due to climate change. Thus, water rationing methods should be revised to improve water allocation efficiency, especially during cyclical scarcity events (i.e., droughts). The proportional rule is the most widely used rationing method to allocate water in water scarcity scenarios. However, this method fails to achieve Pareto-efficient allocation arrangements when there is substantial heterogeneity among water rights holders (i.e., different marginal water productivity) and allocation water markets are narrow or simply nonexistent. In such cases, implementing security-differentiated water rights could improve allocative efficiency during cyclical scarcity periods.

Water management in Spain is affected by the aforementioned circumstances and is thus a good example of a case where the implementation of security-differentiated water rights could improve water efficiency during drought events. The reform of the water rights regime is particularly appealing in Spain given the anticipated ban on water markets. Notwithstanding, the insights obtained from the analysis performed in this paper are also applicable to any country with a mature water economy.

It has been evidenced that security-differentiated water rights are an efficient instrument to share risks related to water supply reliability. This is a growing concern among water rights holders given the increasing uncertainty in water supply due to climate change. Thus, the proposed change in the water rights regime can also be considered as an adaptation measure to climate change, one which is especially suitable when other alternative instruments to manage supply failure risks, such as drought insurance schemes or water options markets, have not yet been developed.

The proposal for reforming the water rights regime in Spain is primarily based on the Australian case study, since this has proved to be the most successful experience worldwide. Moreover, the two countries share some common features, especially relating to their competitive irrigated agricultural sectors that account for more than 80% of total water use. In any case, further research is needed to refine the implementation of the security-differentiated water rights regime proposed here. Indeed, this proposal is just the first step within a longer research project. The next step is expected to involve more in-depth examination in a discussion group including water managers and relevant stakeholders (irrigators, environmental groups, etc.) to fine-tune as necessary the key features (e.g., the percentage of current water rights that should be upgraded into priority rights, the rules guiding the auction procedure, the duration of priority rights upgrade, or the end use of the additional income generated by priority rights). This debate will enable the definition of the operational implementation mechanisms (policy alternatives) for reforming the water rights regime in Spain, which should be ex-ante evaluated using simulation modeling based on mathematical programming techniques. This impact assessment will provide guidelines for policy design aimed at identifying the most suitable option for implementation. Finally, the chosen policy alternative should also be empirically tested in a real-world setting, implementing it as a pilot case study in a Spanish river basin before full-scale implementation at the national scale. This entire procedure will help to guarantee the success of the policy reform proposed.

Nevertheless, it is worth remarking that the insights obtained from the analysis performed and the proposal for reforming the water rights regime are applicable to any country with a mature water economy. Thus, this paper encourages further debates elsewhere regarding how alternative water rights regimes could enhance water management (water rationing) during cyclical scarcity periods.

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Measuring the Transaction Costs of Historical Shifts to Informal Drought Management Institutions in Italy

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Abstract: Coase shows how costly resources are (re)allocated via costly institutions, and that transaction costs must therefore be positive. However, Coase did not elaborate on transitions between institutions which incur positive transaction costs that are characterized by numerous institutional complementarities; that is, feedback loops that inform the need for, and pathways toward, institutional change. Economic investigations of complementary modes of (re)allocation are rarely undertaken, let alone studies of transitions between modes. However, modes of (re)allocation that achieve similar results at less cost are generally viewed as having production-raising value. This paper measures the costs of transitioning drought management institutions in Italy toward informal, participatory, and consensus-based approaches during several recent drought events. The chosen model is Drought Steering Committees, which offer a substitute for current formal (less flexible) planning approaches, and where lower transaction costs that are associated with the transition are inferred. Our results highlight the relevance of empirical assessments of 'costly' transitions based on a historical study of transaction costs, as well as supporting previous works that highlight the value of contextual analysis in economic studies, in order to identify the benefits of institutional investment.

Keywords: Po River Basin; institutional economics; climate change adaptation; cost of adaptation

1. Introduction

Water, an essential resource, is becoming increasingly scarce and costly worldwide [1]. As water scarcity increases, existing institutions that are reliant on inflexible water governance arrangements will constrain corrective action leading to a crisis of governance [2,3]. Identifying or transitioning toward good governance practices and institutions delivering effective, fair and sustainable management of water resources is thus increasingly urgent especially in institutions capable of (re)allocating costly water resources during extreme scarcity events, such as drought.

Generally, institutions can be defined as 'the rules of the game' within which political, social and economic realities operate [4]. Two overarching institutional categories coexist in water resources management: (1) formal institutions, which are established and communicated through channels that are widely accepted as official, such as laws and regulations enforced by authorities and (2) informal institutions, where the social rules, customs, traditions, or codes of conduct are part of the culture and ideology [5]. In both cases, these institution types distribute power to differentially constrain and enable actors and facilitate or limit the response(s) of individuals and communities to climate hazards, such as drought [6]. Further, these institutional approaches may complement and/or substitute for one another depending on governance requirements and choices.

Coase [7] introduced institutional choice to economic investigation, extending a notion proposed by Robbins [8] that transitions between institutions occur within costly frameworks characterized by institutional complementarities. However, although Coase explained that costless bargaining (i.e., zero transaction cost institutions) were unrealistic, the concept of positive transaction costs with respect to institutional substitution was not considered [9]. Ostrom [10], among others, outline ways by which institutional change may be analyzed and selected. However, with respect to transaction cost specifically, while earlier works [11,12] affirm multiple options for dealing with transactions, they do not elaborate upon the role that economic investigation should take in clarifying the function of different modes of resource (re)allocation or organisation. Williamson [13] offers useful insights into governance modes and their selection with respect to economizing objectives (e.g., first order issues to get the institutional environment right, while third order economizing is better aimed at adapting to continuous uncertainty, such as drought). However, Coase typically framed an answer to the comparative institutional analysis problem as one of identifying alternative modes of organisation that achieve similar results at lower costs, which would enable the value of production to increase [11]. An appreciation of these issues by Pagano and Vatiero [9] led them to two hypotheses that we are keen to explore in this paper. The first is that institutional change (i.e., from formal to informal organisation) involves transition and transaction costs, both of which can be empirically measured in order to identify improved (i.e., low(er) costly) governance arrangements (H_1). The second is that costly institutions imply complex complementarities (e.g., feedback loops akin to those discussed by Ostrom [10]), which may limit (promote) substitution. Thus, a historical analysis of the complementary institutional factors framing governance choices will be needed to understand equilibria outcomes (H_2). To test these hypotheses using an applied case study we focus deeply on a set of historical transaction costs and institutional outcomes, which are a key premise of institutional economics.

1.1. The study of Transaction Costs

Transaction costs are defined as the costs of resources used to define, establish, maintain, administer, and change institutions and organizations, as well as those that are needed to define the problems that these institutions and organizations are intended to solve [14]. In the larger context of institutional evolution, they are all of the costs involved in human interaction over time. The arguments for measuring transaction costs represent an increasingly relevant feature in investigations of environmental or common property policy design and analysis, along with their budgets and benefits [15,16].

From an economic perspective, appropriate formal and informal institutional choices include options that minimise/lower all transaction and abatement costs [14]. In the context of complex multiscale problems, such as water management, the measurement of transaction costs usually focuses on markets and other formal institutions [17,18], with little research being conducted on the transaction costs of informal institutions [19]. The latter are frequently used for water resource management in several areas worldwide, particularly to mitigate the adverse effects of droughts, e.g., through informal water markets [20], quota-based water reallocation [21], or risk sharing [22]. Reasons for reliance on informal institutions include trust, networking, shared norms, and reciprocal arrangements, which may help to lower total transaction costs [23].

Measuring transaction costs is challenging, leading Quiggin [24] to describe them as generally being treated by economists as "something of a black box, the contents of which are inaccessible". Most water management institutions do not empirically quantify institutional transaction costs such that they can be easily distinguished from other cost categories. Researchers also report a number of difficulties that are related to the measurement of transaction costs, often suggesting that data are partial and indirect and/or derived from limited cost typologies or proxies to represent transaction costs [25]. Further, there is no broad agreement on a standard terminology about the definition of transaction costs [26]. For this reason, it seems unclear how to identify the peculiarities of a transaction, and which expenses/investment should be regarded as transaction costs. All of the above is even more challenging where informal institutions may amplify accounting data gaps. Consequently, economic investigations of complementary institutional modes of (re)allocation are rarely undertaken while using empirical transaction cost measures, let alone historical studies of transitions between modes.

However, a relatively common feature of transaction cost measurement is the distinction between ex-ante and ex-post costs; that is, those occurring before and after the transaction. The sum of ex-ante and ex-post transaction costs yields total transaction costs. Total transaction costs can be further divided into: (1) administering, monitoring, contracting, and enforcing current policy arrangements (termed *static transaction costs*) and (2) periodically designing, enabling, implementing new, and/or transitioning existing management arrangements to new systems (termed *institutional transition costs*). In addition to these costs, the total transaction costs may be increased when subsequent adaptation requirements are triggered by policy shocks or surprises (termed *institutional lock-in* costs) [14]. Table 1 references the typical transaction costs categories and examples, sub-divided between ex-ante and ex-post transaction costs, which we will focus on later in the analysis section.

Table 1. Categorisation examples of transaction costs, adapted from Garrick [27] and Marshall [14].

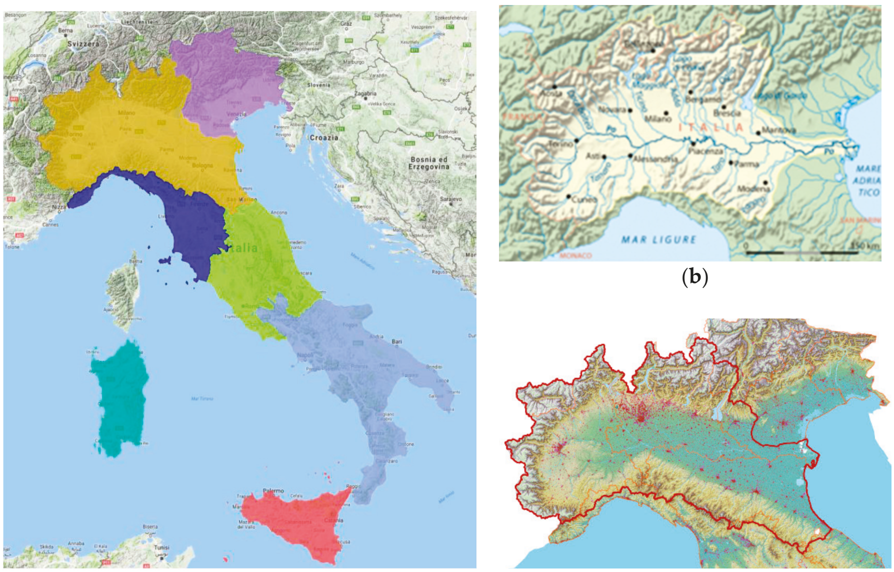
Classes	Sub-Classes	Typology of Transaction Costs	Water Market Arrangement Examples
Ex-ante	Institutional transition costs	Research and information	River Basin development planning and closure (cap on water diversion) Hydrologic and socio-economic studies
		Enactment or litigation	Water rights reform (adjudication, conflict resolution, rules)
		Design and implementation	Modification to storage and distribution, licensing systems and trading rules Water accounting systems
Ex-post	Static transaction costs	Support and administration	Transaction planning, identification of buyers and sellers, administrative reviews
		Contracting	Water rights due diligence
		Monitoring and detection	Water use accounting
		Prosecution and enforcement	Compliance monitoring and enforcement Dispute resolution
	Institutional lock-in costs	Adaptation or replacement	Revised caps on water diversion Adapted water rights and water user association rules Acquiring water rights for the environment if cap on water diversion is revised downward
Source: [28]	Source: [29]	Sources: [14,18,30]	Source: [17,18]

1.2. The Contribution of this Study

The goal of this paper is to evaluate whether, via a case study of informal drought management arrangements in northern Italy, less costly—and ideally improved—governance arrangements have been achieved (H_1). This evaluation will entail a historical examination of the evolution of water governance institutions for Italy, in general, and Po River Basin (PRB) drought management systems in particular (H_2). We will then measure and track transaction costs with respect to transitioning drought management institutions toward informal, participatory, and consensus-based approaches during several recent drought events, with a view to identifying any evidence of low(er) transaction costs coupled to similar—or improved—drought management outcomes. Ultimately, this approach will enable an assessment of the hypothetical propositions and their value for further study to develop the assessment process. The paper is structured as follows: in Section 2, we assess the historical context of the case study area, the PRB in northern Italy; in Section 3, we present methods and data; in Section 4, we conduct an empirical transaction cost analysis of the institutional transition in the PRB; Section 5 discusses the results; and, Section 6 concludes.

2. Historical Institutional Analysis

The PRB is located in northern Italy and extends, with five per cent of its total area (~74,000 km^2), to portions of French and Swiss territory (Figure 1b). In terms of average annual water discharge, the PRB is one of the largest in Europe with an outflow at the mouth of the Po River in Pontelagoscuro of 1470 m^3/s. Po River flow rates depend on the water captured and stored in artificial reservoirs in the mountains, principally in five lakes (Maggiore, Como, Iseo, Idro, and Garda) located at the foot of the Alps. Demand for water is high: the PRB supplies water for hydropower generation in upstream lakes and reservoirs, and potable water to some 3700 municipalities within seven administrative regions with a thriving industry that accounts for 40% of national GDP.



(**a**) (**b**) (**c**)

Figure 1. (**a**) the seven river basin districts in Italy; (**b**) the area of the PRB; and, (**c**) the boundaries of the territory managed by the Po River Basin Authority(red outline).

The system also supplies irrigation water to Italy's largest contiguous agricultural region, which comprises 21.5% of total Italian agricultural land, contributes 30% of national agricultural value-added production [31], and represents around 80% of total water extractions [32]. Water is also needed in the lower reaches of the river to mitigate salinity intrusion during low flow or drought periods—as the area is located below sea-level—and to support fisheries and aquaculture demand.

Average precipitation ranges from a maximum of 2000 mm in the Alpine regions of the PRB to less than 700 mm on the eastern plains, with an annual average of 1100 mm. Under future climate change temperatures will increase, while summer precipitation will likely decrease [33]. Po River discharge is expected to decline during the summer months—when the demand is typically at its peak—and shift to higher levels of discharge in the winter (Figures 2 and 3).

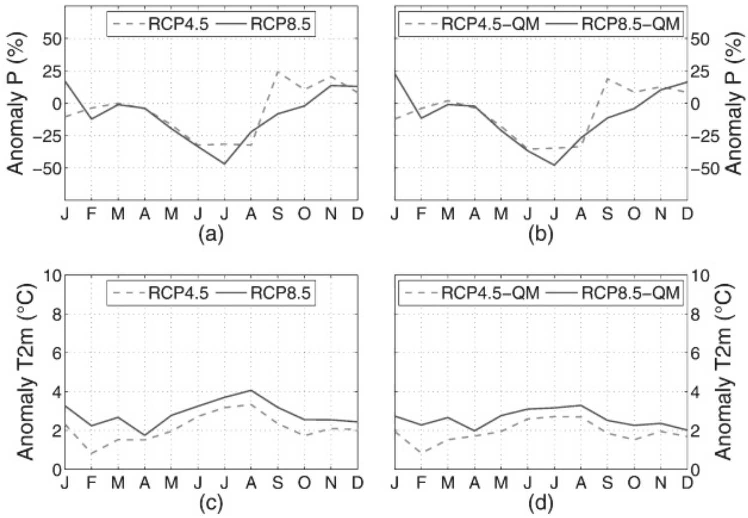


Figure 2. Anomalies in (**a**,**b**) seasonal precipitation in % and (**c**,**d**) two meter mean temperature in °C for the PRB, 2041–2070, versus a 1981–2010 benchmark period. Left side (**a**,**c**) refers to raw CMCC-CM/COSMO-CLM outputs, while the right side (**b**,**d**) indicates the bias-corrected climate projections [33].

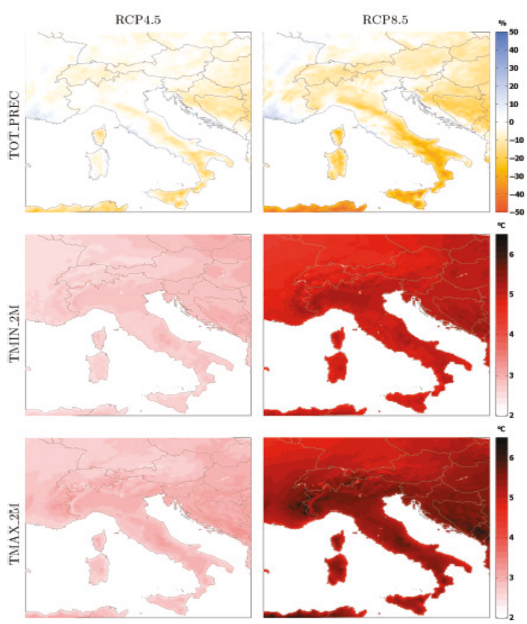


Figure 3. Climate change signal for the period 2071–2100 versus 1981–2010 for mean precipitation, maximum, and minimum temperature [34].

Thus, the frequency and intensity of extreme events, such as droughts, are expected to increase making current levels of water extraction in the basin unsustainable [35]. Evidence of these changes is already noticeable at the regional and local levels, with recorded rainfall reductions and increased temperature variations of around one degree centigrade [36,37]. Droughts also appear to be affecting the region more frequently, with a State of Emergency (SoE) being declared in 2003, 2006, 2007, and 2017. Since 2000, these SoE events have lasted 25 months in total, with an average duration of 6.25 months per declaration. A coordinated climate change adaptation strategy that identifies the main impacts of climate change for a number of socio-economic sectors in Italy was adopted in 2015, followed by a National Adaptation Plan for Climate Change (PNACC) [38]. The PNACC encourages institutions to identify effective ways to mainstream adaptation into existing plans and regulations at different levels of territorial government [21,39]. River Basin Authorities are responsible for identifying and coordinating drought adaptation actions and measures.

2.1. Water Abstraction Licenses Regime in Italy: An Obstacle to Climate Change Adaptation

The current system of creating and managing water abstraction licenses (WAL) in Italy creates a significant obstacle to the effective implementation of these two adaptation strategies. Originally, Italian legislation viewed water as a plentiful resource, and this attitude has remained essentially unchanged since the 1930s. As a result, the volume of authorised WAL in the PRB now exceeds average water availability; for example, current hydroelectric and agricultural licenses amount to 1840 m^3/s, against an average river flow of 1470 m^3/s [21].

Although many licenses are dormant, over-allocation complicates the management of water deficits during drought periods. WAL quotas are also difficult to implement in Italy [40,41] due to the fragmented nature of WAL, and the challenging interplay of Italian water institutions [42] where regional governments have been granted the power to regulate WAL matters. For these reasons, the PNACC proposed a revision of the WAL regime system [38,39]. Recent legal definitions and laws now recognise the limits to national water use, and articulate collective uses of water resources in Italy with respect to protection of environmental water resource uses (Law 183/1989), integrated water resource management (Law 36/1994), and the protection of water quality (Environmental Code 152/2000). The government sought to reorganise water services in the early 2000s, in what was then regarded as a first step towards the introduction of market and pricing reallocation mechanisms. In June 2011, a law favouring privatisation of water supply and sanitation, largely viewed as opening the possibility of water trading, was repealed by referendum. The prevailing view following the referendum was that access to water should be treated as a fundamental right, not subject to free market reallocation. Thus, the referendum outcome limited the use of formal market instruments such as water pricing, trading, or buyback for drought management [43], requiring alternative institutional arrangements. Ultimately, the capacity of river basin managers to coordinate parties and address climate change impacts and future population and economic growth, and/or to prioritise different water uses during drought has been compromised, and regional governments granted the power to regulate WAL matters. Governance of water resources in Italy thus remains complex, emergency-driven, and focused on short-term problem-solving. This is particularly evident during drought events in 2003, 2006, 2007, 2015, 2016, and 2017, where reactive strategies probably increased the negative impacts of water scarcity.

2.2. Formal Drought Management Institutions

In the absence of market-based reallocation mechanisms drought management in Italy has traditionally focused on formal command and control approaches, where the state intervenes in the management of basin water resources as a last resort instrument (Law 225/1992) to enact water restrictions with sanctions for non-compliance [44]. By contrast, recent evidence of climate change and increased drought events from 2003 onwards have served to focus EU Member States' attention on alternative political and technical responses that involve participatory (e.g., informal) approaches [45] over prescriptive (e.g., formal) sanctions. A key document was the communication addressing the problem of water scarcity and droughts in the European Union [46], which presented an initial set of non-mandatory policy options at the European, national, and regional levels to address and mitigate the challenge posed by water scarcity and drought.

During the process of transitioning the European Water Framework Directive (EU-WFD) into national legislation, the PRB experienced a severe drought event in 2003 that presented a significant threat to urban, industrial, and agricultural water supplies. The Italian government formally declared a SoE, which enabled them to: (i) centrally manage drought emergency interventions in the PRB for a period not exceeding 180 days (but which could have been extended by another 180 days by the central government); and, (ii) allocate funding for initial drought management interventions, with the option for further interventions where recognised as necessary by the delegated commissioners in charge of managing the emergency. This formal institutional arrangement was managed by the National Civil Protection Department (NCPD), anchored to the Presidency of the Council of Ministers which supervised all activities.

2.3. Informal Arrangements for Drought Management—The Case Study

In the 2003 drought event, the NCPD and Po River Basin Authority (PRBA) jointly sought to avoid last resort interventions by the central Italian government. Both were concerned about the impact of the drought on energy supply, and the need to act more rapidly (and collectively) to address issues in line with EU best drought management practices. Consequently, a Drought Steering Committee (DSC) was initiated, presided over by the PRBA, with the purpose of coordinating communication and voluntary responses to drought across a large number of organisational members. The DSC constituted an informal institution, because it was not legally recognised, and stakeholders participated on a voluntary basis. Further, there was no capacity for sanctions in the case of non-compliance with decisions made at the meetings and, in cases of conflict, the DSC could not be sued and/or prosecuted due to its informal status. Therefore, any decisions had to be made via agreement or consensus due to a lack of explicit legislative (formal) mandate in support of those activities. Ultimately, trust among the membership, networking and shared objectives were expected to reduce total institutional transaction costs of drought management, as outlined below.

The mission of the DSC, sanctioned under a Memorandum of Interest (MoI), was to manage severe water deficits in a unified manner and to delay or prevent critical water shortages. Two main objectives were included in the MoI: (i) maintenance of minimum water withdrawal opportunities for downstream irrigators and Po River Delta water users (e.g., aquaculture); and, (ii) maintenance of hydroelectric outflows to guarantee maximum possible electricity production, as requested by the national transmission grid operator. Under these common objectives, the DSC initiated a network of information gathering aimed at measuring lake storage data, monitoring of PRB water flows in real time, and a summary of WAL water uses. These measures served to better assess and understand the negative impacts of the drought, contributed to an overall stabilization of water flows and availability, and brought progressive increases in supply to WAL-holders during the drought. This initial success meant that, since 2003, the DSC has been convened again when necessary to deal with PRB drought events and to limit (potentially costlier) state intervention. Drought management planning through the DSC is now enshrined in the Po River Basin Plan [47], along with requirements for water-stress mapping, temporary restriction measures for intensive (e.g., back-to-back rotation) cropping, and early-warning systems that are based on basin modelling.

The success of the DSC has also become a reference point for the management of water crises in Italy more generally, given its capacity to aggregate and coordinate various stakeholders' interests when considering regional differences. Therefore, the DSC is now recognized by the Italian government as an effective instrument for the fair and sustainable management of water withdrawals. In 2016, legislation provided for the mandatory activation of a DSC in each of the seven Italian basin districts, along with responsibility for coordinating different local water authorities. These DSCs are aimed at harmonizing adaptation efforts under the larger Permanent Observatory (PO) institutional structure in

Italy, which monitors climate dynamics and variability, climate hotspots, and natural environmental hazards from extreme weather events.

The success of the original PRB-DSC suggests that it may provide a useful model for jurisdictions beyond Italy, particularly in the EU. Incentives for a jurisdiction to participate in their own version of the DSC are two-fold. First, the DSC represents an opportunity to coordinate with other water users before any drought declaration is made, after which centralized (distant and/or coercive) decision-making arrangements may dominate to reduce negotiation/adaptation opportunities. Second, the DSC is an opportunity to foster greater mutual understanding and trust among relevant organisations, increased information exchange, and collaboration between water users that may otherwise be hampered by administrative and political fragmentation. The informal nature of the DSC may also lead to relatively inexpensive institutional arrangements that are more readily enacted (institutional transition costs) and administered (static transaction costs) by other watersheds with limited or poor water right structures.

From this assessment, we conclude that our understanding of the equilibria transition from formal to informal drought management institutions in the PRB is enhanced by considering complementarities and how they have hindered certain institutional choices, while fostering the selection of others. This lends support to H_2 and the value to economic investigations from a consideration of the historical context. However, whether the transition has broadly resulted in low(er) costly modes of organisation (H_1)—and therefore productivity increasing outcomes—is the subject of our subsequent analysis.

3. Materials and Methods

3.1. Stakeholders, Interviews, Document Analyses, and Assessment of Governance Arrangements

Our measurement of transaction/transition costs was based upon extensive stakeholder consultations. The stakeholders are all of the interested parties who affected, were affected by, or otherwise influenced drought governance decisions. We defined the domain of stakeholders involved in the DSC and different focus levels, which range from identifying relevant institutions and key persons to finding the interactions and associated transaction costs. Our methodology comprised: (i) analysis of water allocation governance frameworks in place; and, (ii) analysis of informal DSC institutions and how these are embedded within the national and regional PRB governance. Initial meetings were held with senior members of the DSC to identify whom to interview. Face-to-face and telephone interviews were scheduled and conducted involving a total of 12 experts, with each interview lasting around two hours. The interviews enabled us to explore technical and organizational details that are necessary to identify sources of transaction cost data.

3.2. Transaction Costs Data Collection, Categorisation and Analysis

McCann et al. [30] established a framework and typology for transaction costs measurement based on previous work from Thompson [48], which we follow in this study. The data collection approach is similar to that detailed in Loch and Gregg [49]. The main function of the DSC is to coordinate stakeholder participation and consensus in the wake of significant drought event periods. Routine technical meetings during non-drought periods—which, together with hydrologic basin modelling, constitute the bulk relevant transaction costs—are also commonly arranged by regional authorities with the support of Environmental Protection Agencies (EPAs). DSC meetings were used to track stakeholder involvement, with the salary cost rates (per hour) at each expert-level providing a proxy base value for transaction costs estimates. These data were obtained while also considering: physical or virtual participation by experts in meetings; estimates of travel distances and/or costs from the organization to which they belong to the venue of the meeting; and, the duration of the meeting. Information for the study was collected through interviews and meetings minutes. For some meetings the minutes were not available, requiring additional interview data collection to fill information gaps. Our approach was informed by previous studies that interviewed government staff [50] and representatives of stakeholder groups [51] to identify the time spent on various relevant activities within the organisations. Further, in 28 out of 235 cases, the mean salary cost values (~€70,000 per annum) had to be assigned when information was not publicly available or provided in the interviews. DSC meetings and related transaction costs were then classified based on their key focus: meetings to agree memoranda of understanding involved ex-ante enactment costs; meetings to develop/test new hydrologic models for the basin involved ex-ante design and implementation costs; meetings to extend the modelling framework and, thus, enhance institutional capacity to monitor water use and compliance and limit illegal abstractions that are involved ex-post monitoring and detection costs; while meetings to incorporate the DSC institution within the PO arrangements for Italy as a whole provided some measure of lock-in (i.e., substitution-hindering complementarity) transaction costs.

The DSC was assisted by the PRBA through organisation of meetings, data collection and analysis, and technical advice. Initially (2003–2008), this role was accomplished with the support of an external service provider that was subsequently transferred to the PRBA (2008–2016). Financial data from the PRBA provided transaction costs related to the collection of information in support of decision-making by the DSC, including hydrologic modelling and analysis. As an example, two external staff from the Regional Environmental Agency of the Emilia-Romagna Region (ARPA-ER) worked part-time on the development and maintenance of the hydrological model to support DSC activities. It should be noted that the total transaction costs involved in the DSC process were absorbed by different organisations at different points of the original program life-cycle (2003–2016).

Table 2 summarizes for the case of the DSC the classes, sub-classes, typology, and categorisation of transaction costs, plus the data sources used for data collection.

Table 2. Categorisation of transaction costs, adapted from [30], Garrick [27], and [14], including categorisations identified for the Drought Steering Committee (DSC) case study, and related data sources.

Classes	Sub-Classes	Typology of Transaction Costs	Categorisation of Transaction Costs for the Drought Steering Committee	Data Source
Ex-ante	Institutional transition costs	Research and information	The meetings of the DSC (minutes)	Meeting minutes (stakeholders involved, duration of the meeting), personal interviews (salary cost rates, physical or virtual participation, participation in meetings, travel distances, duration of meeting) and estimates through sensible adjustments of comparable costs (travel costs)
		Enactment or litigation	Enactment: includes all the meetings for the signing of the memorandum of understanding for the DSC	
		Design and implementation	Hydrologic studies and modelling of allocations supporting the decision of the DSC	Financial records and other publicly available information (reports)
Ex-post	Static transaction costs	Support and administration	The organisation of the meetings (design costs)	2003–2008: Financial records; 2008–2016: Structured interviews with representatives of stakeholders to obtain information of the personnel involved, plus estimates through sensible adjustment of salary costs
		Contracting	Not present	NA
		Monitoring and detection	The meetings for the hydraulic modelling	As in research and information typology
		Prosecution and enforcement	The meetings of the PO	As in research and information typology
	Institutional lock-in costs	Adaptation or replacement	Meetings to include DSC arrangements within PO framework	As in research and information typology
Source: [28]	Source: [29]	Sources: [14,18,30]	Source: Authors' elaboration	

As an example, in order to calculate the research and information costs corresponding to the physical participation of an expert from Torino in a DSC meeting, the travel time between Torino and Parma (headquarters of the PRBA) was obtained, and multiplied by a standard cost per km to generate the transportation costs by car, or alternatively the cost of the train ticket was used, depending on the type of transportation used. This amount was added to the salary cost rate (per hour) times the duration of the travel plus the duration of the meeting to obtain the corresponding transaction cost(s). Following this travel cost calculation, we could estimate that a representative of the Regional Environmental Agency of the Piedmont Region (ARPA-Piedmont), taking part in an in person meeting in 2017, spent EUR 120 in the train trip (economy ticket, high speed train). Next, the salary cost was obtained from institutional salary tables (60,000 EUR/year), its' hourly equivalent calculated (assuming a standard 36 h/week working time and 52 weeks per year yields EUR 32.1), and multiplied by the duration of the meeting (1.3 h) plus the duration of the round trip (5.2 h), which gives as a result EUR 208.3. The total cost for this participant is therefore estimated at EUR 328.36 (208.3 + 120).

Another example is provided for hydrological model implementation, the most significant transaction cost in the 2006–2011 period. This transaction cost is obtained as the sum of the cost of the contract with an external provider during the 2006–2011 period, obtained from accounting records (EUR 700,000), plus the cost of the personnel employed by ARPA-ER from 2008 to support the consulting firm and maintain and update the model once the consultancy was over, which is obtained as in the example above multiplying the hourly cost of the personnel dedicated to model support and maintenance times their dedication to the task.

After data for each cost item were carefully collected and calculated, they were transformed into real values using 2017 as the base year (e.g., meeting costs during the 2003 drought were converted into euro of 2017 using data from the World Bank [52]).

All final transaction costs were then categorised into institutional transition (ex-ante) and static transaction (ex-post) costs, as per Table 2. Following the method adopted by Loch and Gregg [49], analyses were performed to identify: trends in each category over time, summed total transaction costs for the DSC, and comparisons between drought and non-drought periods. The following sections detail the results of the institutional mapping exercise, which assists in our assessment of whether the institutional transition achieved similar/improved outcomes, and subsequent transaction cost analysis to measure and assess the costs of that process.

4. Results

4.1. Stakeholder Map and Assessment of Drought Governance Arrangements

Current drought management systems in the seven Italian river basin districts involve three main actors with differentiated roles and responsibilities for River Basin Management Plans (RBMPs): national government and ministries in coordination role; river-basin district authorities in operational role; and, regional governments and administration in both coordination and operational roles (Figure 4). They are all part of PO, and they have to implement the RBMP through a Protection Plan (PTA) by addressing the qualitative and quantitative water resource management objectives.

Based on the objectives of the PTA, the Optimal Territorial Areas (ATO, for the domestic use of water) and the Land Reclamation Boards (LRB, for the management of irrigation water) are in charge of preparing the Area Plan (AP, in Italian: *Piani d'Ambito*) and Water Conservation Plans (WCP), respectively. During this process, drought is monitored through the relevant sub-basin's Drought Management Plan (DMP), a subsidiary instrument to the RBMPs that assesses the basin status on a continuous basis using four stages (normal, pre-alert, alert, and emergency), and identifies appropriate measures for delaying and/or mitigating drought impacts (e.g., information campaigns) [40]. Therefore, a variety of legislative requirements must be adhered to with respect to drought events. Critical Italian government institutions (from 2016 onwards) include the NDCP, the Ministry of Agriculture, the Ministry of Infrastructure, and the Ministry of Environment; all of which are accompanied by the National Association of Land Reclamation Boards (ANBI), the Italian research organization dedicated to the agri-food supply chains (CREA), the National Institute of Statistics (ISTAT), Institute for Environmental Protection and Research (ISPRA), the Foundation representing companies operating in the public services of water, environment and energy (UTILITALIA), the Association for the reorganization of the Integrated Water Service (ANEA), and the National electricity company association (ASSOLETTRICA). The PO are now operating in each of the seven Italian RBDs: Padano (i.e., PRB), Alpi Orientali, Appennino Settentrionale, Appennino Centrale, Appennino Meridionale, Sardegna, and Sicilia. The PRB regions are the Autonomous Region of Valle d'Aosta; Piedmont; Liguria; Lombardy; Emilia-Romagna, Veneto; Autonomous Province of Trento; and, Toscany.

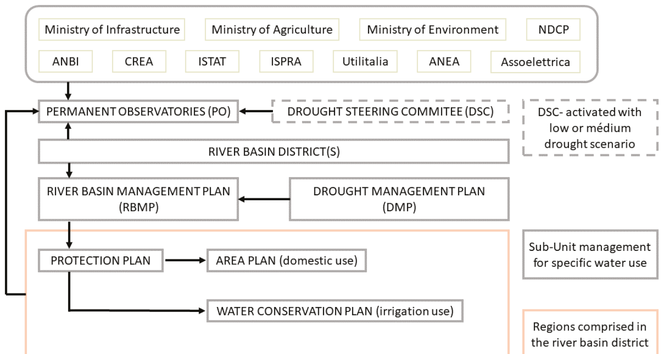


Figure 4. Framework of drought management planning and arrangements in Italy.

When a drought emergency is declared in the PRB, the DSC is triggered. Naturally, this process requires coordination at a decentralized level. The PRBA is responsible for coordinating all DSC stakeholders (local and national), and their responses to the emergency drought status (Figure 5). The PRBA collects, updates, and disseminates information on the availability and use of water resources across the relevant river basin organisations. These include: the Italian Ministries of Agriculture, Environment, Infrastructure, and Productive Activities; representatives from each of the five Lake Regulators; the Dam Management Agencies; the operator of the national transmission grid (GRTN); the inter-regional agency for the Po river (AIPO); the national Association of Land Reclamation Boards (ANBI); the agencies responsible for energy supply (SPE); representatives from regional drought committees responsible for managing these emergencies at the local level; and, a representative from the autonomous province of Trento. The PRBA is responsible for notifying these stakeholders that a DSC has been convened, and inviting them to participate in the process and provide the latest technical synthesis reporting to describe current water resources through indicators, bulletins, reports, etc. This technical information is supported by hydrologic modelling data and technical information provided by ARPA-ER, and used to reach decisions on water reallocation via agreement or consensus.

From the interview process, it became clear that, when first implemented, the DSC was not trusted to deliver interventions on its own and needed the administrative support from one or more relevant authorities (i.e., the PRBA and other key institutional stakeholders). However, this is changing under new PO regulatory structures aimed at strengthening informal cooperation and dialogue between water governance organisations within each district to promote sustainable use of water resources in line with the EU-WFD. Nevertheless, these arrangements did not increase formal institutions. The PO is a voluntary and subsidiary structure supporting integrated water governance to manage the collection, update, and dissemination of data on the availability and use of water resources in the districts. Thus, the PO provides guidelines rather than prescriptive arrangements for the regulation of withdrawals, resource use, and possible compensation to users. During droughts, the PO interacts with the DSC to ensure common objectives that include an adequate flow of information that is necessary for the assessment of critical water scarcity levels, the evolution of that scarcity and current water withdrawals, and for implementing appropriate emergency actions to proactively manage the drought event. Therefore, public and private organizations at all levels of water governance can participate in the decision-making to achieve these common strategic objectives during a drought.

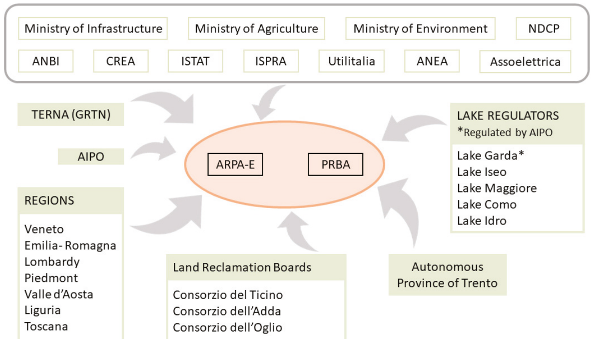


Figure 5. Participatory map of the Permanent Observatory (PO) of the PRB and stakeholders, 2016–ongoing.

Thus, the arrangements identified for the PRB above offer a good example of informal water governance institutions for managing drought events, where we recall that: (i) the DSC is not legally recognized and stakeholders participate on a voluntary basis; (ii) there is no capacity for sanctions in the case of non-compliance with decisions made at the meetings; and, (iii) in cases of conflict, the DSC cannot be sued and/or prosecuted because of its informal status. Yet, the arrangements detailed above also have an increased potential to meet EU-WFD objectives over existing institutional approaches due to their integrated water resource management methods, coupled with processes aimed at avoiding political or legal interference (last-resort measures) during drought emergency response implementation. The DSC demonstrates capacity for coordinating actions on a voluntary basis and encompassing a wide range of stakeholder trust (democratic legitimacy), while achieving robust water governance institutions. Thus, the transition to informal institutional arrangements in support of successful adaptation to drought events appears to have achieved improved drought management outcomes, but at what cost?

4.2. Transaction Costs Measurement and Analysis

We must be able to observe some reduction in the average static transaction costs and that any periodic institutional transition costs associated with drought events must be short-lived (i.e., evidence of improved total outcomes) in order to test whether a transition to informal institutions with improved outcomes has been achieved at low(ered) costs over time. Our measurements of total DSC transaction costs for establishing, coordinating, and managing the DSC are summarised in Figure 6, while the share of ex ante and ex post transaction costs is shown in Figure 7—where a change in (ex-post) transaction costs for new institutions cannot take place without (ex-ante) transition costs in support of those changes. A more detailed breakdown of the individual ex ante and ex post transaction cost categories is available in Appendix A. The base-line for our cost-reduction analysis is the 2003 drought event, when the DSC officially came into existence.

The initial transaction costs were relatively significant in that year, consisting mainly of enactment and research/information gathering investments. Growth in total transaction costs was then experienced in response to three-consecutive drought events (2005–2007). This corresponded to investments in further information gathering, administrative costs for the DSC, and hydrological modelling to monitor water use across the relevant PRB sub-regions. Interview analysis revealed that a significant fraction of these costs that are involved identifying and agreeing upon common objectives for the DSC, consistent with informal network requirements and building trust between the stakeholders.

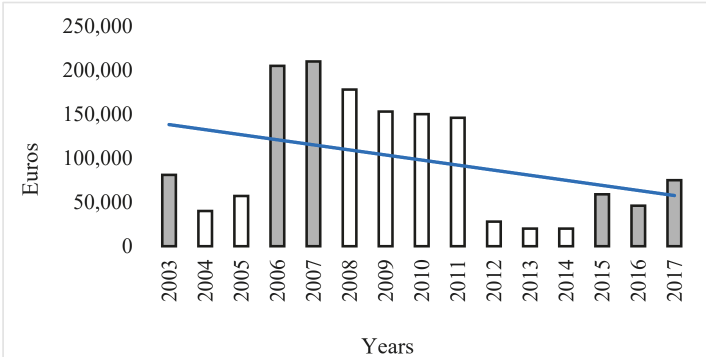


Figure 6. Total transaction costs for the DSC (years with droughts are in grey).

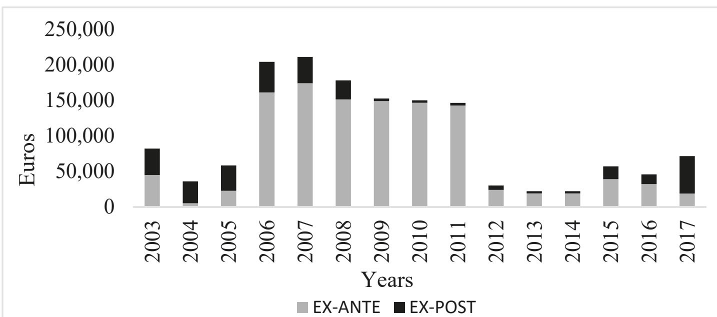


Figure 7. Ex-ante and Ex-post transaction for the DSC. Droughts in 2003, 2006, 2007, 2015, 2016, 2017.

Post-2007, no drought emergency events occur in the PRB. Investments in the hydrological modelling continued at high levels for a few years (2008–2011) until the contract with the external provider that supported the development of the model finished. After 2011, the DSC total transaction costs generally fell due to reduced hydrological modelling implementation costs and because extraordinary meetings were not needed; thus administration costs for routine management comprised the majority of required investment. However, in the period between 2015 to 2017 the PRB experienced a series of consecutive drought emergency events. This period also reflected a shift toward interaction with the PO arrangements, requiring some increased transaction costs. In response, the total transaction costs rose over that period due to increased administration and the enforcement of DSC requirements—but critically this increase is approximately one-third of the peak transaction costs of previous periods. Some of that lowering of transaction costs was due to an increased use of technology to support/conduct DSC meetings, as well as a lower degree of drought severity in the later events, relative to the period before 2010. Many of the meetings were now held at the PRBA while using media (Skype) lowering the requirement for travel and salary costs to attend meetings in person for many of the organisations, as well as the response and coordination times for managing drought emergencies.

With specific regard to individual transaction costs categories (Appendix A, Figure A1), the average static transaction costs decreased over the period considered, while short-lived institutional transition costs increases were observed during drought events (Figure 6). In total, the trend is downward, which suggests a lowering or minimisation of total costs across the life of the informal DSC governance arrangements.

According to Garrick [27], such trends indicate robust institutional outcomes—i.e., institutions that are capable of taking corrective action through “relatively less transaction cost-intensive autonomous and planned adaptation". For our purposes, the measurement of transaction costs enables a confirmation of positive transition costs to establish new institutions—as we should expect, and in support of H_1—but also that this new mode of organisation provides scope for productivity and efficiency gains for Italian water users.

5. Discussion

The results from our analysis of the collected data offers a novel contribution to the transaction cost literature by: (i) applying ex-ante and ex-post transaction cost measurement to informal water governance institutions, (ii) providing evidence in support of the usefulness of measuring transaction costs for evaluating institutional transition or substitution objectives, (iii) highlighting the relevance and value of historical context for economic investigations; and, (iv) showing how informal institutions may underpin water governance/management arrangements to lower total transaction costs related to drought management in an EU context. Beyond our support for the two main hypotheses, the results from the informal management of drought events at river basin scale determined the following key points.

5.1. Drought Management Arrangements

Drought requires a flexible management approach that is able to monitor the evolution of the event, to then respond within and across multiple governance levels (e.g., across multiple economizing orders in Williamsons' framework [13]). In comparison to formal arrangements that are available in Italy, the informal DSC approaches outlined above may be more flexible and adaptive with respect to drought management and adaptation (third-order economizing), which is also consistent with new EU water governance objectives. Shifting the management focus to a local level increases the appreciation of drought impacts, and provides for more appropriate responses in shorter timeframes than that of monocentric models, although such shifts may also lead to local capture of, and rent-seeking in, the policy process.

Positive effects of the DSC also arise from improved information transmission among stakeholders, and a tangible capacity to lower drought impacts and increase adaptive capacity. Further, monitoring the availability of water resources (inflows, reservoirs, outflows) and their adjustment in real time has allowed for the DSC to more quickly recognise and react to drought events via the use of short to medium term forecasting tools, drought indicators, and event evolution scenarios. These scenarios have also contributed to the construction of regional technical tools in support of managing water balances at the basin scale. Finally, the recent institutionalisation of DSCs and relevant stakeholder involvement across all (ordinary) periods of water management through the PO, rather than limiting their existence to drought periods, is an improvement upon the typically reactive (emergency) commencement of Italian management measures.

Without a measurement of the marginal centralised transaction costs in contrast to counterfactual institutional arrangements, we cannot draw any formal conclusions regarding the value for money or total transaction cost differentials. However, the PRB DSC arrangements have now been extended across each of the seven River Basin Districts (RBDs) in Italy, formally established in May 2017. According to interviewed stakeholders, the DSC arrangements were attractive to the Italian government because they did not require any additional funding to implement (i.e., lower transition costs), while avoiding some negative impacts of drought events (i.e., improved management outcomes). Thus, it seems logical to conclude that the political value of these transaction costs and their institutional outcomes has been recognised. By favouring an informal institution, like the DSC, the Italian government could potentially observe an increase in the effectiveness of water governance arrangements, although it will require further evidence over time to support this conclusively. This will be the focus of a future research project involving hydro-economic modelling of costs and benefits.

5.2. Transaction Costs and Policy Performance Analysis

Our findings are relevant for policy makers and other stakeholders beyond the PRB. Here, the measurement and analysis of transaction costs undertaken paves the way toward performance assessment of similar initiatives based on informal voluntary partnerships for water management in Italy and Europe. These include incipient river contracts, forums for dialogue and knowledge sharing between public/private stakeholders, and local communities in compliance with the EU's subsidiarity principle, which are gaining momentum in Italy and elsewhere in Europe [53]. A constraint to any application of the findings reported here may arise from the non-conjunctive catchment characteristics of the PRB; that is, they do not share water resources with other basins. This is often not the case for the other river basin contexts in Italy or elsewhere in Europe, for whom the issues may be more challenging as a consequence, and involve higher transaction costs.

Moreover, comparisons of the cost-effectiveness of alternative policy options to enhance flow rates during droughts must account for the total costs of the options relative to a baseline or status-quo scenario. These include the transaction costs of the reform measures, along with any abatement costs incurred by economic agents during the implementation of local adaptation strategies. Recent research focusing on the analysis of abatement costs in the PRB shows that the proportional rule used to reallocate water under the DSC approach—which relinquishes a fixed percentage of the initial allocation from users, irrespective of the economic losses involved—underperforms other formal drought management arrangements, such as water charges [54]. This gap will be further amplified via forward and backward linkages among economic sectors within the PRB, and with other Italian regions outside the basin. Thus, a complete policy performance assessment calls for empirical analyses that combine transaction and abatement costs estimates [55]. This too will be incorporated into future research work in the area.

5.3. Transaction Costs and Uncertainty Analysis

Finally, water resource management is performed in a context of Knightian or deep uncertainty, where it is often not feasible to identify all of the possible outcomes and/or assign a probability to each identified possible outcome [56]. Under deep uncertainty, rather than optimal institutional settings, we should aim for robustness through the avoidance of path dependent institutional trajectories to enable future adaptation in the face of unpredictable future events that are explainable only after they happen. This requires adaptive institutional frameworks [27].

As indicated above, our transaction cost measurement framework can provide initial information on the robustness/adaptive ability of PRB institutional arrangements. However, conclusions regarding the robustness of these arrangements in response to future uncertainty would need to consider additional measures of adaptive efficiency according to Garrick [27]. For completeness, these measures would also have to include the lock-in cost impacts of institutional options to allow for a cost-effectiveness evaluation [14]. Similar to the work undertaken by Loch and Gregg [49], this would entail identifying and measuring three performance indicators over space and time: (1) how well the drought management objective(s) have been met; (2) the average transaction costs per unit of those met objective(s); and, (3) total program budgets. For adaptively efficient and robust institutions, these three performance indicators should be increasing, decreasing and sufficient respectively. Measures of these indicators are beyond the scope of this pilot study, but remain an objective for a wider research program focused on identifying instruments best-suited to achieving water policy and management targets. The wider research focus of this work will examine maximised benefits per unit of transaction cost (alternative measure of cost effectiveness), as well as maximising the net public/private gains from transaction cost expenditure (social welfare). This broader assessment framework should enable a more comprehensive assessment of total policy or program benefit-cost outcomes.

Finally, future climate change and economic dynamics may change the outcomes that are reported in this study. Further research will be necessary to determine under what conditions this may happen, and any requirement to adjust or change policy accordingly [57].

6. Conclusions

Transaction costs matter for effective organisation and institutional management of scarce and costly resources, such as water. During times of drought, formal institutions may provide costly and inflexible management arrangements that may increase the total transaction cost requirements. This paper explores the transaction costs that are associated with a historical transition toward informal drought management arrangements in the PRB of northern Italy. We test two hypotheses related to the value of transaction cost analysis in support of institutional transition/substitution choices, and the value of historical context to economic investigations. By measuring and tracking transaction costs with respect to drought periods in the basin we explore the total costs associated with a new institutional approach, and note that the DSC arrangements have been mandatorily adopted by the six other River Basin Districts in Italy—somewhat ironically, as this has formalised what was originally an informal process. It remains to be seen whether the formalisation of drought management arrangements based on the PRB DSC will ultimately increase total transaction costs, or further reduce the total transaction costs of drought management in Italy by following a participatory, consensus-based approach elsewhere. However, it is impossible to draw more robust conclusions without a more detailed study of centralised costs. That said, in contrast to standard approaches where a complete set of empirics might be provided, some may find our approach here less satisfying. However, we would argue that value is provided by the thought and measurement processes that have gone into the study, rather than arriving at any 'number'. The process of empirically identifying, measuring, and assessing transaction costs is in its infancy; but remains a critical means by which adaptive effectiveness and efficiency for future institutional choices will potentially be explored, as we have done in this case. While our empirics may not be complete they do provide a valid contribution where—as we have pointed out—it is our intention to explore additional means by which we can get at a final set of 'numbers' in support of the full costs and benefits. Like all good research, it is a process, and one that we are interested to continue following. Overall, though, our study highlights the usefulness of transaction cost case studies, and the need for extensions to this approach that incorporate not only transaction and abatement cost minimisation evaluations, but also assessments of per unit private/public welfare benefits that accrue from policy and programs, such that more comprehensive evaluations and uncertainty analyses may be achieved in the future. We believe this to be a rich area of future research that may require the incorporation of climate, hydrological, and economic modelling assessments to be successful.

Author Contributions: Conceptualization, S.S., A.L., and C.D.P.-B.; methodology, A.L., S.S.; validation, S.S.; formal analysis, S.S., A.L., C.D.P.-B.; investigation, S.S.; resources, S.S., J.M.; data curation, S.S.; writing—original draft preparation, A.L., S.S., C.D.P.-B.; writing—review and editing, A.L., C.D.P.-B., S.S.; visualization, J.M.; supervision, A.L., C.D.P.-B., J.M.; project administration, J.M.; funding acquisition, J.M., A.L., C.D.P.-B. All authors have read and agreed to the published version of the manuscript.

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Conflicts of Interest: The authors declare no conflict of interest.

Appendix A

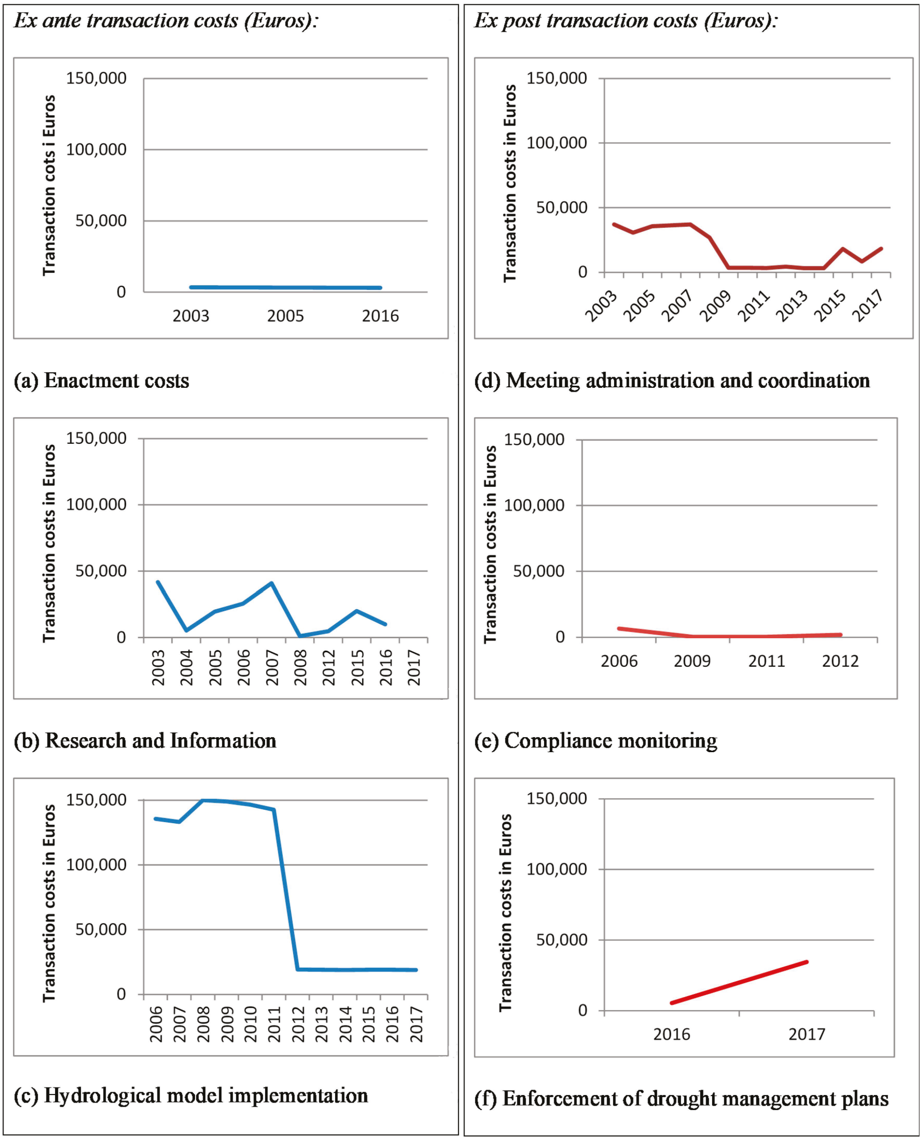

Figure A1. Measures of DSC individual ex ante/ex post transaction cost categories over time.

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Review

Water Markets in the Western United States: Trends and Opportunities

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Abstract: Efforts to address water scarcity have traditionally relied on changing the spatial and temporal availability of water through water importation, storage, and conveyance. More recently, water managers have invested heavily in improving water use efficiency and conservation. Yet as new supply options become harder to find and/or appropriate, and demand hardens, society must consider other options to, if not reduce scarcity, minimize the impacts of such scarcity. This paper explores the role water markets are playing in addressing water scarcity in the American southwest: a water-limited arid and semi-arid region characterized by significant population growth rates relative to the rest of the US. Focusing on three representative southwestern states—Arizona, California, and Texas—we begin by highlighting how trends in water supply allocations from different water sources (e.g., surface water, groundwater, and wastewater) and water demand by different water users (e.g., agricultural, municipal, and environmental) have changed over time within each state. We then present recent data that shows how water trading has changed over time—in terms of value and volume—both at state level and sector level aggregates. We end with a discussion regarding some institutional adjustments that are necessary for water markets to achieve their potential in helping society address water scarcity.

Keywords: drought; water markets; Western US

1. Introduction

One of the most pressing challenges confronting the US in the 21st century is water scarcity. Population growth, which will increase the demand for water throughout the US, has risen by nearly 7%, or approximately 22 million people, since 2010 (Figure 1). Of course, there have been increases in water use efficiency that have somewhat counteracted the impact of population growth on demand. For example, in California, per capita daily use dropped from 244 to 178 gallons from 1995 to 2010 [1]. Yet increased evapotranspiration from a warmer climate suggests a less available supply reaching our municipalities, agricultural lands, and water bodies, likely increasing the level of conflict among water sectors. Furthermore, while most climate change models suggest that the amount of precipitation may not change significantly over the next 50 to 100 years, precipitation events will become much more variable, intense, and infrequent, with more precipitation falling as rain than snow [2–5]. Combined, these characteristics suggest that conflicts over water scarcity will increase as the temporal distribution and form of supply deviates from what our infrastructure was designed to handle.

The objective of this review paper is to shed light on how water scarcity is changing in the Southwestern US, and the role water markets have and might play in addressing this scarcity.

In particular, we focus on how the demand and supply of water are trending in representative states in the southwest—Arizona, California, and Texas—and the increasing role of water markets in helping states to address such scarcity. Two of these states—Texas and California—have accounted for nearly 1/3rd of the population growth in the US (Figure 1). Relative to US averages, the southwestern states of Arizona, California, and Texas confront higher population growth (2.45% vs. 1.15% between 1920 and 2018), higher temperature (61.1 °F vs. 52.5 °F), and less precipitation (20.68 in vs. 30.48 in). Such differences increase water demand and decrease the supply of runoff from precipitation events resulting in rising water scarcity.

Since markets depend on differences in the marginal values across users to create incentives to trade, we differentiate between different types of water use (e.g., agricultural, environmental, municipal/city) to better understand which sectors will likely be driving the market, where scarcity might arise within a state, and the role of water markets in potentially assuaging such scarcity. After briefly describing some general climate and population statistics within each state that likely influence water scarcity, we introduce water supply and demand conditions by state, with a brief background of water use trends.

Following each state-level discussion, we provide data on water market trends and transactions within each state and discuss how those trends may relate to water scarcity characteristics within each state. Note that the effectiveness of water markets and growth in water demand, supply, and use is largely influenced by each state's water rights laws and regulations. Given space limitations, we have opted to focus strictly on presenting the most recent data on water demand, supply, and markets but direct the reader to other sources for an in-depth understanding as to how water rights and regulations within each state influence the trends we identify. For example, for California, see Hanak, et al. [6]; for Arizona, see Colby and Isaaks [7]; for Texas, see Kaiser [8].

Data are presented on the overall market size measured in total volume and value during 2009–2018 as well as the distribution of market activity across western states. We also review active sectors buying and selling water and discuss commonly traded types of water entitlements and transaction structures. In this paper, we use water markets data from Waterlitix™, the largest and most comprehensive database of water rights price and sales information in the United States. Waterlitix™ is a proprietary database developed and maintained by WestWater Research. The data are the results of two decades of continuous, primary research of water right trading and leasing. Transaction information is compiled from state and local regulatory filings, public and private transaction documents such as leases and purchase and sale agreements, and through direct interviews with parties involved in transactions. The database is structured to include both water asset/water right details and transaction specific information. Water asset information includes details on the water asset type involved in the transaction such as authorized diversion volume, quantity of water approved for transfer (which may differ from the authorized diversion volume), other information on the water rights or assets such as priority date, authorized use, source and locational characteristics including water basin, administrative districts or water management boundaries such as a water district or ditch company. The database also includes specific transaction details such as buyer and seller information, previous and new use of the water, transaction structure such as single year lease, multi-year lease, permanent purchase or other complex exchanges where financial consideration is paid. Other transaction information includes financial consideration paid, financial and transaction terms, total payment, and unit price payment that has been normalized across all transactions to allow for comparisons of equivalent transactions and water asset types. All of the transactions within Waterlitix™ are geo-referenced within a geospatial searchable data platform. Prior water market studies include comprehensive transactions from 1987 to 2009 in the Western US [7,9–12]. Our analysis provides an update to these prior studies. We end with a discussion of how the role of water markets may be improved in the future to help states, and the US as a whole, better cope with future rising water scarcity.

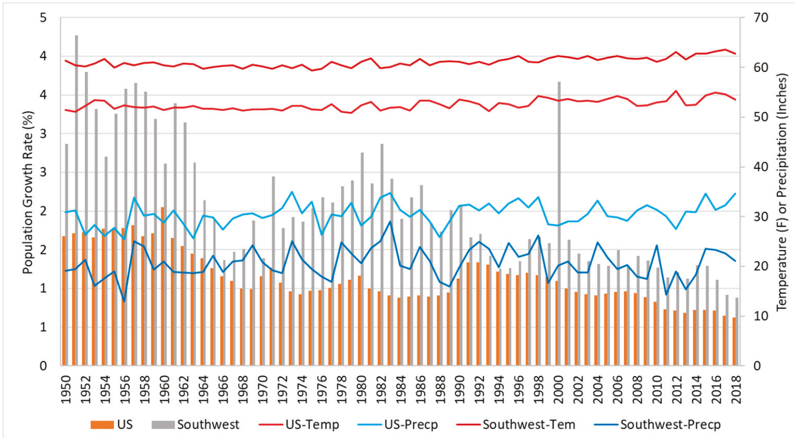


Figure 1. Changes in population over time, average annual temperature, and total precipitation in the US and southwestern states (Arizona, California, and Texas) (1950–2018). Source: Authors calculations, US Census Bureau for the population estimates, and National Oceanic and Atmospheric Administration (NOAA) National Centers for Environmental Information for temperature and precipitation [13]. Notes: Population growth indicates annual population changes in the US and average annual population changes in the three states included in this review. Temperature and total annual precipitation indicate the average annual statistics for the US and the three states.

2. State-Level Water Summaries: Trends and Trades

In this first section, we provide a brief discussion of general water scarcity conditions in each state, and both state and sector water demand, supply, and market trends.

2.1. Arizona

Arizona encompasses a variety of landscapes, ranging from desert to mountain, within an arid to semi-arid climate. Average annual precipitation varies from around 40 inches in mountain areas in the east-central part of the state, down to approximately 3 inches in the southwest region, which is comprised of a hot desert landscape (with temperatures in the summertime between 105 and 115 °F) [14]. Responses to these challenges, though, have led to Arizona having one of the most progressive water management systems' in the southwest. Much of Arizona's water comes from the Colorado River, which meets approximately 32% of the state's surface water withdrawals. With the looming pressure surrounding an over-allocated Colorado River, Arizona must tackle issues of diverting water supplies into rural communities while managing its limited supply. Rural communities, in particular, are more significant in the context of Arizona's water supply issues due to inadequate groundwater and few surface water rights, high population growth rates, in combination with water supplies that often are vulnerable to drought, and limited hydrogeological information from these areas [15–18].

2.1.1. Water Supply and Demand

In 2017, the four primary sources of water in Arizona included groundwater (1.44 million acre-feet, MAF), Colorado River water (1.22 MAF), other in-state surface water supplies (0.84 MAF), and wastewater (0.23 MAF). Using the most recent available data in 2017, groundwater was the leading supplier to the agricultural sector (0.85 MAF) and the industrial sector (0.17 MAF), while the Colorado River was the main supplier to the municipal sector (0.55 MAF). Effluent supplies are allocated to municipal (0.11 MAF), industrial (0.09 MAF), and agricultural (0.03 MAF) sectors. As shown in

Figure 2a, Arizona has increasingly been relying on water supplies from the Colorado River and local effluent since 1985, while local surface water supplies are declining.

Over 50% of Arizona's total supply of around 3.75 MAF is allocated to agriculture [19]. In addition to agricultural demand, there is a significant—second only to agriculture—demand by the municipal sector to keep pace with Arizona's growing population. Native American and industrial water demand round out the other two significant categories of demand, the latter of which is largely influenced by the US demand for copper, of which Arizona supplies 65%. Note that Native American water supply and demand in this article refers to Indian reserved water rights. Arizona has many Indian reservations, both on the Colorado River and in central Arizona, close to Phoenix and Tucson [20]. The mining industry, on average, uses about 96,200 acre-feet annually to run its operations and generate power for its plants.

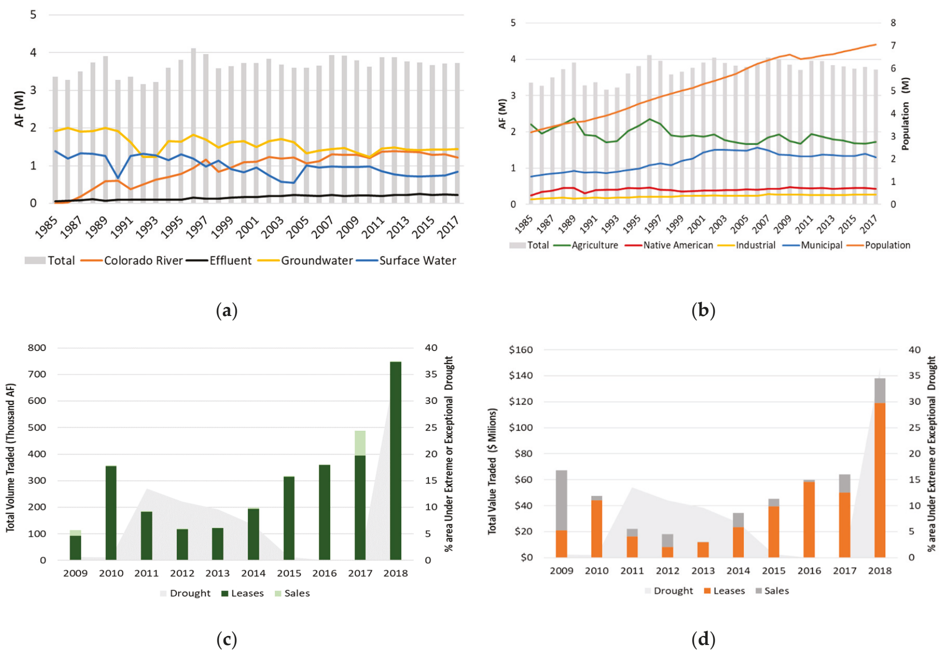


Figure 2. (**a**) Water supply by source, (**b**) water demand by sector, (**c**) total volume traded, and (**d**) total value traded in Arizona. Source: Authors calculations, Arizona Department of Water Resources (DWR), and WestWater Research. Drought data are from the US drought monitor [21]. All prices are in real 2009$ using the Consumer Price Index (CPI)—All Urban Consumers Average from the Bureau of Labor Statistics (BLS).

As mentioned above, demand for water in Arizona primarily comes from the agricultural and municipal sectors, followed by Native Americans and industry. As shown in Figure 2b, we see that while Arizona's population has grown by nearly 100% since 1985, overall water use has increased by only around 10%. Over this period, demand for water by the agricultural sector has been on a slight decline, while the municipal sector, which saw significant increases in water use due to population growth and development in the late 1980s through the early 2000s, has tapered off.

2.1.2. Water Trading

Arizona has an active water trading market. From 2009 to 2018, nearly 151,000 acre-feet (AF) of water was traded annually (Figure 2c), which comprises approximately 4% of its overall consumptive water use annually. There has been a near seven-fold increase in total volume traded since 2009, with a clear trend upwards since 2012. During the extreme or exceptional drought years of 2010 to 2015, traded volume was relatively low compared to 2018, which was also considered an exceptional drought year. In terms of types of trades, 92% of the water trades were in the form of leases, while only a small volume (~8%) was in the form of permanent sales. Noteworthy, in 2018, approximately 20% of the total water supplied was associated with some water trade.

Figure 2d juxtaposes the trade value (in 2009$) over the past ten years across Arizona with the percentage of Arizona under extreme or exceptional drought [21]. In 2018, the over 375,000 acre-feet of water traded through leases and sales had a total market value of $138 M (million) (in 2009$). Market activity has been increasing significantly since 2013, an attribute that may also be related to a healthier US economy, which experienced a significant downturn in 2008, beginning with the housing crises. While it seems apparent that the market is responding to the drought of 2018, both in terms of value and activity, this contrasts with the drought from 2011 to 2014 in which trading activity in the form of a lease or purchase prices did not seem to respond.

Columns 2–4 in Table 1 show the acre-foot price of water leased or sold during the last ten years as well as percentage area within the state under extreme or exceptional drought. As indicated, the price associated with permanently traded water was around $2046/AF, on average, whereas the price associated with a temporary sale registered at approximately $130/AF, on average; consequently, the lease price was about 6% of the sales price. Looking at the change in the three-year moving average over 2009–2018 indicates that the price per acre-foot traded through leases increased slightly by 1.40%.

Table 1. Leases and sales price United States Dollar/acre-feet (US$/AF) by year (2009–2018) and state.

	Arizona			California			Texas		
Year	**Leases**	**Sales**	**D3–D4 [1]**	**Leases**	**Sales**	**D3–D4 [1]**	**Leases**	**Sales**	**D3–D4 [1]**
2009	228	2125	1	224	1544	2.08	96	4217	16
2010	125	807	1	197	2498	0.00	122	3293	1
2011	89	2252	14	183	5981	0.00	106	501	68
2012	69	3067	11	224	3692	0.34	115	3016	26
2013	99	1131	10	218	3797	6.64	112	4290	22
2014	121	2032	7	334	9230	75.37	186	1903	17
2015	126	1796	1	446	3700	70.19	159	793	5
2016	162	1294	0	381	4095	48.48	164	1354	0
2017	126	153	0	278	2707	1.84	163	1119	0
2018	159	5806	37	287	5442	2.35	167	3023	6
Average	130	2046	8	277	4268	21	139	2351	16

Notes: All prices are in real dollars in 2009 using the CPI—All Urban Consumers Average from the BLS [22].
[1] Average annual percentage area under extreme (D3) or exceptional drought (D4) [23].

2.2. California

Two characteristics that define California are climate variability within the state and the geographic mismatch between the sources of supply and the bulk of demand. That is, average annual precipitation varies from less than 5 inches in the arid to the semi-arid southern part of the state to more than 100 inches in the more mountainous northern parts [24]. This characteristic also leads to the challenge that over 1/3rd of its water supply comes from northern California, while the bulk of demand, from agriculture to the large urban centers in and around Los Angeles, is from the central and southern parts of the state. As such, water conveyance, storage, and transfer are very much ingrained into California's development path, factors that are critical to changing the spatial and temporal availability of water in California, and the ability of water trading to complement its water portfolio.

2.2.1. Water Supply and Demand

Based on data from 2001 to 2015, the three primary sources that comprise the nearly 61 MAF of water supply in California include surface water (60%), groundwater (22%), and wastewater (18%).

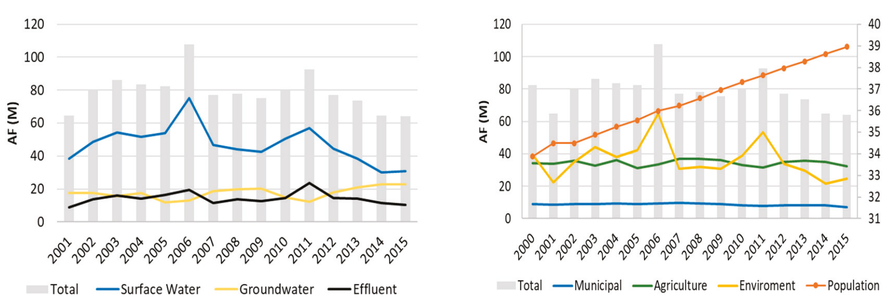
Comparing 2015 to 2001, on average, surface water supplies decreased by 1%, groundwater supply increased by 2%, and treated wastewater supplies increased by 1% (Figure 3a). These estimates are largely influenced by the severe and extreme drought California experienced starting around 2013 that resulted in reduced surface flows and aquifer overdraft.

On the demand side, approximately 89% of California's water goes to environmental and agricultural usage and the rest to the urban sector. Environmental water use refers to water in rivers to protect "Wild and Scenic", instream flows to maintain habitat, water to manage wetlands, and water to maintain urban and agricultural water quality (i.e., Delta outflow) [25]. Figure 3b illustrates the trends in water demand by sector as well as population growth in California since the 2000s. Comparisons among sectors indicate that the average annual growth of water demand from 2000 to 2015 in the urban sector was slightly negative (−1%), as was the growth in the agricultural sector (−0.33%). On average, water allocated to the environment was down by approximately 2%. Interestingly, even though the California population grew significantly between 2001 and 2015, overall urban water use declined, primarily due to efficiency and conservation measures enacted by Californians, including during the drought in 2014 to 2016. Similarly, improvements in irrigation efficiency facilitated the downward trend in water use by the agricultural sector.

2.2.2. Water Trading

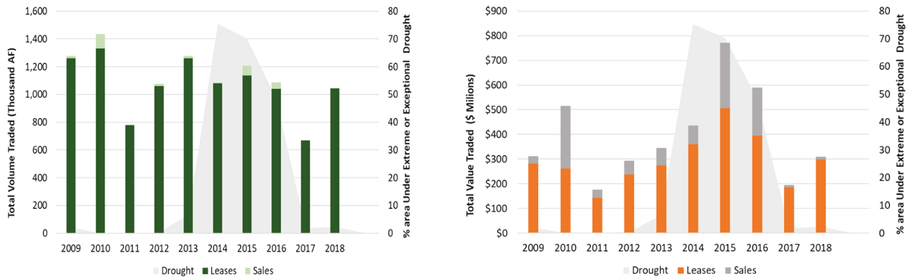
California's water markets are comprised of transferring rights either in the short-term (less than one year) or long-term (greater than one year). The majority of California's water rights are held by the farm sector, which has the majority of water sales primarily in California's San Joaquin Valley. More recently, lease activity has increased, dominating the market share in terms of traded volume and value. In terms of the average annual volume (Figure 3c), from 2009 to 2018, nearly 1.1 MAF of water was traded in the form of leases and slightly over 29,000 AF in the form of permanent sales. Given California's overall annual water allocation is around 61 MAF, water trades account for around 2% of the supply, having decreased slightly over the past decade.

Figure 3d illustrates how the total value of water trades have changed over the past ten years. As shown, during the height of the most recent drought, the value of sales soared up to nearly $800 million in 2015, dropping precipitously to nearly $300 million in 2018 after the drought subsided. In the past two years, the number of permanent sales decreased significantly, with nearly 79% of the trade value tied up in leases. In terms of water prices, columns 5–7 in Table 1 indicate that the price of an acre-foot of leased water reached its apex in 2015 during the worst period of the drought, which is also when the price of permanent water also reached its highest level (over double its ten-year average). In terms of prices, leases, on average, sold for around $277/AF, while permanent sales sold for around $4268/AF. As expected, prices tend to increase during periods of significant drought.



(**a**) (**b**)

Figure 3. *Cont.*



(**c**) (**d**)

Figure 3. (**a**) Water supply by source, (**b**) water demand by sector, (**c**) total volume traded, and (**d**) total value traded in California. Source: Authors' calculations, California Water Plan updates [26], and WestWater Research. Drought data are from the US drought monitor [27]. All prices are in real 2009$ using the CPI—All Urban Consumers Average from the BLS.

2.3. Texas

Water laws and policies in Texas are continuously changing in order to accommodate the growing population and demands while adjusting to changing climate and drought. The state's primary abundance of resources, such as cattle, agriculture, and oil are dependent on the water supply in a state with significant climate variability. Precipitation varies from around 9 inches, on average, in the west and southern part of Texas to approximately 60 inches in the east and northern parts. The temperature varies between 16 °F and 50 °F (with an average of 32 °F across the state) in January to between 88 °F and 100 °F in July (with an average of 94 °F). While average statewide precipitation of around 27 inches may seem significant, the overall demand for water based on predicted population growth is projected to increase by up to 22% by 2060. This population growth, when coupled with climate change and other factors contributing to drought, including increased evaporation and ground absorption, presents significant challenges to Texas in its efforts to confront water scarcity. Challenges include an estimate water shortage of 8.9 MAF annually in 2070, caused by current supply allocation problems [28].

2.3.1. Water Supply and Demand

The water supply resources in Texas emanate primarily from two sources: groundwater and surface water [29]. As illustrated in Figure 4a, groundwater, which comprises approximately 54% of the state's overall supplies, has been decreasing over the past two decades, dropping from around 10 MAF to around 7.5 MAF. Surface water, which comprises nearly 43% of the state's overall supply, has experienced some variability over the past two decades but has generally remained slightly below 6 MAF. Recycling has contributed a minor amount to Texas's overall water supply portfolio. The overall decline in available water supplies in Texas nearly mirrors the decline in aquifer storage.

Similar to other states, the major diverter of water in Texas is agriculture, which uses approximately 60% of the state's overall supply, followed by the municipal sector, which uses approximately 1/3rd of the overall water. While municipal water demand has slightly increased since the 2000s, agricultural water use has declined somewhat significantly, from slightly less than 10 MAF in 2000 to slightly less than 8 MAF in 2017 for an approximate 20% reduction. Industrial use, approximately 1 MAF per year, has trended slightly downward as well. According to a water usage summary report for 2017 conducted by the Texas Water Development Board (TWDB) [29], municipal water is primarily sourced from surface waters, approximately 64%, while groundwater supplies municipalities with approximately 32% of its needs, with the remaining 4% coming from effluent (Figure 4b).

2.3.2. Water Trading

In Texas, groundwater trading is much more prominent compared to Arizona and California. For example, approximately 69% of the total value traded in Texas between 2010 and 2014 came from Edwards Aquifer, an active market for sales and leases of groundwater entitlements [30]. From 2009 to 2016, the volume of water traded in Texas increased annually up to nearly 240,000 AF, which is about 2.4 times the amount that was traded in 2009. For 2017 and 2018, there was a slight drop to approximately 200,000 AF annually. Given that there is approximately 13 MAF used annually in Texas, trading accounts for less than 2% of this usage. As Figure 4c shows, there was a spike in permanent sales during the height of the drought in 2011, but otherwise traded volumes mostly occurred through leases.

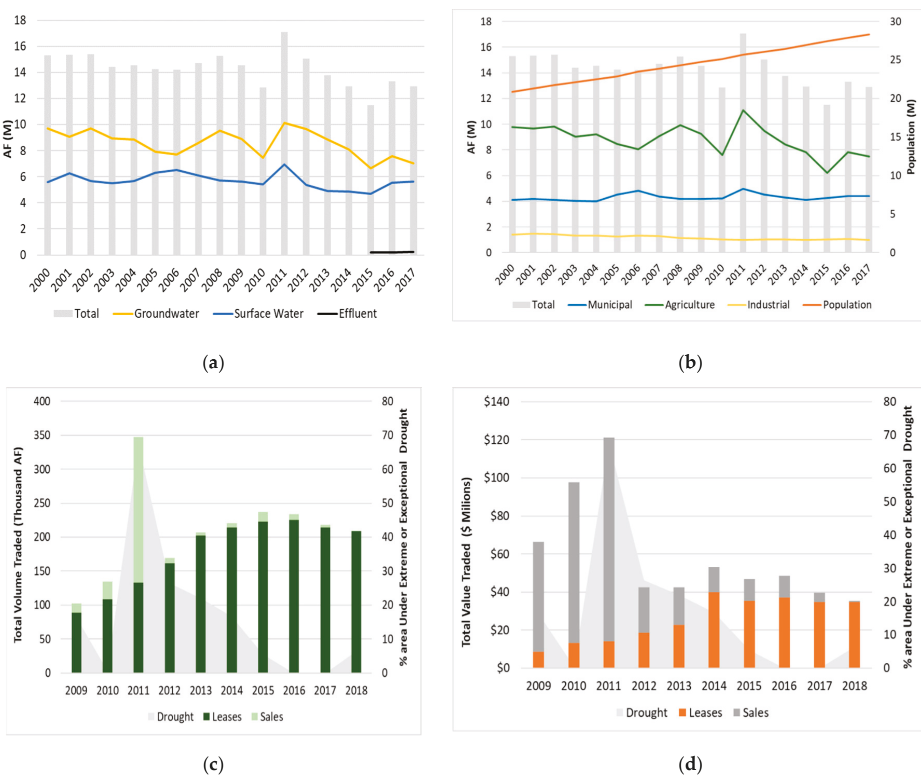


Figure 4. (**a**) Water supply by source, (**b**) water demand by sector, (**c**) total volume traded, and (**d**) total value traded in Texas. Source: Authors' calculations, Texas Department of Water Resources, and WestWater Research. Drought data are from the US drought monitor [31]. All prices are in real 2009$ using the CPI—All Urban Consumers Average from the BLS.

In terms of value, we see quite a different story. The years 2009 to 2011 saw the highest value in water trading over the past ten years, with 2011 reaching nearly $120 million in sales (primarily due to permanent water sales). The trading value decreased quite significantly from 2012 to 2018, with permanent sales decreasing significantly (Figure 4d). Interestingly, in considering columns 8–10 in Table 1, we see that while the price per unit of permanent water transfers and leases was highest in 2012 and 2013, the volume traded was low, as was the overall value, especially relative to 2011 which experienced significantly lower prices but higher volumes.

3. Discussion

In this section, we first provide a comparison between the three states in terms of water demand and supply by sector and source. We then provide a discussion of notable water market characteristics—both similarities and differences—across the three states. We conclude with a brief discussion on the importance of developing more transparent and efficient markets to facilitate the usefulness of this tool in helping states confront rising water scarcity.

As illustrated in Table 2, across all three states, the agricultural sector requires the highest volume of water, with it consuming 58% of the water in Texas, 51% in California, and 46% in Arizona. Note that while we use the term "consuming," a more accurate term would be "diverting" since a fraction of the water not transpired or evaporated often returns to the system [32]. However, while agriculture consumes the most water, its overall use has gone down over the past two to three decades, most notably in Arizona (~22%) and Texas (~12%). Improvements in irrigation efficiency are responsible for much of this decline. On the municipal side, demand is trending slightly up in Texas, is somewhat stable in Arizona, and trending slightly downward in California over the past two decades, with water efficiency measures again playing a significant role in counteracting significant population growth in all three states. As the agricultural and municipal sectors adopt more efficient water use behavior and technologies, demand hardening (i.e., as farmers/households become more efficient, it becomes more difficult to further reduce demand during a shortage or drought) will ensue thereby increasing the potential benefits of water markets as a tool to address increased future water scarcity.

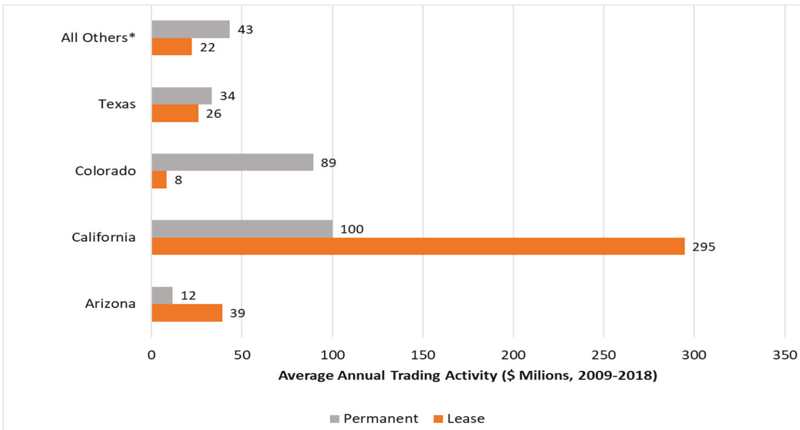
Table 2. Comparison of demand share for each sector and supply share from each source in percentage terms across the three states.

	Demand			Supply		
	Agriculture	**Municipal**	**Industrial**	**Surface**	**Groundwater**	**Effluent**
Arizona [1]	46.28	35.02	7.26	55.22 [2]	38.73	6.05
California [3]	50.55	10.92	-	47.97	35.78	16.25
Texas [1]	58.01	34.22	7.77	39.01	54.44	1.58

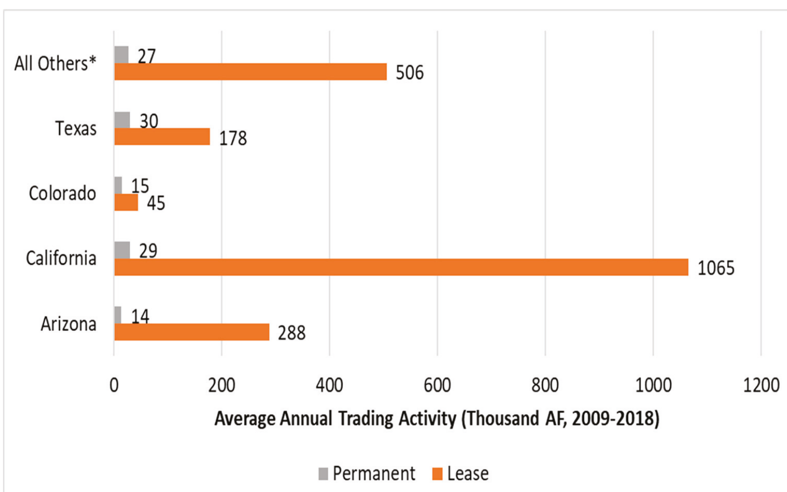
[1] Using the most recent available data in 2017. Arizona also includes demand by Native Americans (11.44%). [2] Sum of surface water share (22.51%) and water from the Colorado River (32.71%). [3] Using the most recent data in 2015. Demand-side in California also includes demand for the environment (38.53%).

In terms of source supply, surface water is the primary provider in Arizona (55%) and California (48%), but in Texas, groundwater is the primary source (54%). Unlike the role the Colorado River played for Arizona during the 1980s and 1990s, surface water sources are unlikely to provide any new volumes to these states moving forward, and groundwater supplies are in decline in all three states. Furthermore, with the enactment of new groundwater sustainability legislation in California in 2014, groundwater pumping is likely to decrease even more than it currently is. As such, water markets again can play an increasingly important role in responding to an increased level of water scarcity due to declining or less reliable water supplies in each state. Of course, effluent in the form of treated municipal wastewater may help assuage such scarcity in local markets, as is taking place in California, yet such efforts require significant infrastructure investments, along with other costs due to technological constraints [33,34], to play a more significant role.

Since 2009, our data suggest that water markets in all three states are functioning to help address water scarcity, although there is likely plenty of opportunities for improvements and growth. While California has by far the highest amount of water trading—both in terms of volume and value—Arizona's market transactions comprise approximately 4% of the overall water used in the state, double the approximate 2% that defines both California and Texas (Figure 5a,b). However, even at 2%, approximately $3.9 billion of water was exchanged in California over the past decade. Most of the activity and value is derived through temporary leases rather than permanent sales, on average. Exceptions to this include significant increases in the price of permanent water sales in California during its most recent drought, although trading activity did not change significantly, and permanent sales and the value of those sales in Texas during the drought in 2010 and 2011. Overall trading activity, though, has been on the rise in Arizona, somewhat stable in Texas, and quite variable in California over the past decade, highlighting the importance of the heterogeneous market, environmental, and institutional conditions across states, conditions that determine market performance.



(**a**)



(**b**)

Figure 5. Average annual trading activity by state and trade type (2009–2018). (a) Average annual trading activity ($ Milions, 2009-2018); (b) Average annual trading activity (Thousand AF, 2009–2019); Source: Authors' calculations and WestWater Research. Notes: * All others include other western states, including Idaho, Montana, Nevada, New Mexico, Oregon, Utah, Washington, and Wyoming.

In terms of who is selling and/or leasing the water, Figure 6 provides a comparison of the sources of supply and demand for water trades and transfers in California (Figure 6a,b), Texas (Figure 6c,d), and the Western US (Figure 6e,f) across the major sectors. In California, agricultural water rights holders provided most of the water to the market over the past ten years (Figure 6a). Approximately 76% of the total volume transacted over this timeframe originated from the agricultural sector, followed by the municipal sector (19%). Agriculture's market share as a supplier has increased over the last ten years by around 2%, although it should be recalled that overall traded volumes have gone down slightly.

Figure 6. Summary of water trade activity by sector in California, Texas, and Western US (2009–2018). (**a**) Volume sold by sector (1000 AF) in California; (**b**) volume purchased by sector (1000 AF) in California; (**c**) volume sold by sector (1000 AF) in Texas; (**d**) volume purchased by sector (1000 AF) in Texas; (**e**) Volume sold by sector (1000 AF) in the Western US; (**f**) volume purchased by sector (1000 AF) in the Western US. Source: Authors' calculations and WestWater Research. Note: The western states include Arizona, California, Colorado, Idaho, Montana, Nevada, New Mexico, Oregon, Texas, Utah, Washington, and Wyoming. Note that we did not have state-level data on Arizona.

In terms of major buyers in California, the municipal sector purchased/leased the most water, on average, over the past ten years, followed by agriculture and then the environment (Figure 6b). While the municipal sector had a somewhat stable level of purchases from 2009 to 2016 (on average they comprise approximately 55% of the total market share), there was a slight decrease in 2017 and 2018 potentially due to (i) lower incentives to trade due to drought subsiding in 2017, and (ii) increased acreage of higher revenue perennial (e.g., tree and orchard) plantings whose significant capital investments increase the opportunity cost of fallowing land. While the agricultural sector has comprised approximately 25% of the market purchases over the past ten years, there was a noticeable and significant increase in the year 2018, as the drought eased and groundwater regulations were tightened under the recently passed Sustainable Groundwater Management Act of 2014. The environment, meanwhile, also plays a significant role in California water markets, comprising approximately 18% of total transactions by volume traded.

In Texas, similar to California, agricultural water rights holders provided most of the water to the market over the past ten years (Figure 6c). Approximately 89% of the total volume transacted over this timeframe originated from the agricultural sector, followed by the industrial sector (8%). In terms of major buyers, the municipal sector purchased/leased the most water, on average, over the past ten years (84%), followed by agriculture (13%) (Figure 6d).

The Western States in this figure include Arizona, California, Colorado, Idaho, Montana, Nevada, New Mexico, Oregon, Texas, Utah, Washington, and Wyoming. Not surprisingly, water is sourced primarily from the agriculture sector. Over the past ten years, approximately 73% of the total volume transacted in the Western US originated from the agricultural sector (Figure 6e). The municipal sector is the second-largest supplier, comprising approximately 23% of the overall sourced supply in the Western US. The industrial sector is responsible for approximately 4% of the sourced water. Note that while in Texas the agricultural sector is responsible for typically around 90–95% of the overall water sales/leases in the state, in absolute terms it is relatively minor compared to California, where the municipal sector is responsible for around the same volume of water sales/leases compared with the agricultural sector in Texas.

On the demand side, participation remains relatively stable, with municipalities continuing to be the largest buyer with 46% of total market share over the past ten years (Figure 6f), although it plays a much more significant role in percentage terms in California and Texas (Figure 6b,d). Environmental buyers, usually comprised of private entities, conservation groups, but also state and federal agencies in efforts to maintain or meet obligations associated with environmental quality, instream-flows, and wildlife habitat [35,36], also play a significant role in western water markets—mostly arising in California—comprising approximately 26% of total transactions by volume traded, followed by agricultural (16%) and industrial (12%) sectors. As noted in Szeptycki, Forgie, Hook, Lorick, and Womble [35], there is significant variation in how water transfers for the environment are regulated across western states, and these differences can significantly limit the type and scope of transfer. While all three states we considered have opportunities to reduce obstacles that are hindering environmental transfers, particularly the administrative burden buyers and sellers confront exercising such transactions, California and Texas are noted to confront fewer of the legal challenges than Arizona in terms of the scope, certainty, and permissibility of environmental water transfers.

In considering the year-to-year variation, we see that there were some significant volumes purchased by the agricultural sector in Texas during the drought years between 2011 and 2014, but on average, over 90% of the volume bought was by the municipal sector. While purchases of permanent water or leases by industrial users do happen, the percentage of the overall volume is quite small. Finally, and what perhaps California's experience in 2018 forebodes for the rest of the West, water supply firming for agriculture associated with the increase in permanent cropping, especially in California, has prompted agriculture to participate in the demand side in higher proportions. As shown in Figure 6b, California's agriculture demand-side market participation has increased by 6% and 15% by value and volume traded, respectively, over the last ten years.

4. Concluding Remarks

Water markets at their core, as with any market, are intended to help reduce the impacts of scarcity by facilitating the transfer of water to its highest-valued uses. What this review has shown in evaluating three western states, is that water scarcity is likely to increase significantly moving forward, primarily due to population growth and the added water demand associated with such growth. Of course, improvements in water use efficiency, both in the agricultural and municipal sectors, have helped society respond to date (indeed, overall water use in California has decreased in the agricultural and municipal sectors). However, demand will harden, and thus such efficiency gains will be harder to come by, resulting in water demand rising with population growth. Scarcity will also heighten due to lower and/or more variable supplies coupled with increased regulation surrounding groundwater pumping and use. These conditions, increasing demand coupled with stagnating or declining and more variable supplies, which seem to characterize each of the three states we examined, suggest an increasingly important role for water markets.

Our analysis has also shown how water markets have played an essential role in water reallocation throughout the Western US. In the recent data we analyzed here, most of the transfers are associated with leases as opposed to permanent water sales. Nevertheless, the overall amount of water that is transferred is small relative to the total water used, between 2% and 4%. This suggests that plenty of opportunities exist for the market to expand, which will require attention from market developers, regulators, and stakeholder input. For instance, during California's most recent drought, trading activity did not seem to respond in any appreciable manner, yet the price of both leases and permanent sales rose significantly (e.g., the price of permanent sales rose from $3797 per acre-foot in 2013 to over $9230 per acre-foot in 2014, yet actual trading activity in the state declined). There are multiple factors —that differ across states— that likely contribute to inhibiting the market from achieving its full potential, including high transaction costs associated with often multiple layers of approval, a lack of transparency, poor and incomplete information flows, along with conveyance and infrastructure limitations.. So while markets have been serving as a means to help change the temporal and spatial distribution of water allocations to their higher-valued uses, significant opportunities exist to both better understand the drivers that influence water market performance and expand the market through the creation of a more transparent, flexible, and user-friendly system.

While water transfers can lead to an overall increase in the net benefits water use from a social perspective, concerns of third-party effects and externalities on other users can create challenges and limit the full functioning of a water market [37]. For instance, if water transferred out of a region results in impacts on local employment and income, such third-party effects can lead to transfers being politically unattractive (and lead to limits on transfers). Of course, if the transfers occur within a particular region, then such third-party effects will be minimal. In response to these third party effects, governments often respond by limiting out-of-region transfers via mandates or fees. Alternatively, if transfers incentivize greater groundwater pumping in agricultural-based communities, this may have impacts on the availability of municipal water for those communities dependent on groundwater for health and hygiene [38]. Careful hydrological monitoring, or employment of a general water accounting framework, can help policy makers better understand the potential implications of transfers on groundwater levels and other users.

Note that the "water market" we describe in this paper is comprised of significantly different water trading and transfer schemes both within and across the three states analyzed. While our focus was on traded water entitlements, surface and groundwater rights are the most commonly traded asset class within the Western US market. However, there are other types of ownership interests in water that are also traded. For example, in Arizona and California, groundwater banking is also traded. Entitlement to use treated wastewater is traded in Arizona, California, and Colorado. Entitlement to store water for use in a surface reservoir, known as "Storage Water Rights", is observed in California and Colorado [39]. As such, there are many opportunities and forms of markets that can be used to help the Western US cope with rising water scarcity, but it requires significant planning, cooperation, collaboration, and evaluation by policymakers with stakeholders to facilitate the development and implementation of such markets.

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Review

Hydro-Economic Modelling for Water-Policy Assessment Under Climate Change at a River Basin Scale: A Review

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Abstract: Hydro-economic models (HEMs) constitute useful instruments to assess water-resource management and inform water policy. In the last decade, HEMs have achieved significant advances regarding the assessment of the impacts of water-policy instruments at a river basin or catchment level in the context of climate change (CC). This paper offers an overview of the alternative approaches used in river-basin hydro-economic modelling to address water-resource management issues and CC during the past decade. Additionally, it analyses how uncertainty and risk factors of global CC have been treated in recent HEMs, offering a discussion on these last advances. As the main conclusion, current challenges in the realm of hydro-economic modelling include the representation of the food-energy-water nexus, the successful representation of micro-macro linkages and feedback loops between the socio-economic model components and the physical side, and the treatment of CC uncertainties and risks in the analysis.

Keywords: hydro-economic modelling; water policy; climate change; river basin management

1. Introduction

Population growth and economic development constitute the main forces behind processes such as irrigation expansion, urbanization, and industrialization, all of which trigger increasing water demands and therefore water scarcity as well as water stress, both in terms of water quantity and water quality [1–4]. Climate change (CC) may act as an amplifier of these impacts on water resources [5]. Water scarcity also constitutes an economic problem and has become a serious limitation for socio-economic development worldwide [6]. The gap between water demand and supply capacity that exists in many parts of the world leads to higher competition between alternative uses (and economic sectors). Water scarcity and extreme events exacerbate this competition for water resources and generate negative social and economic impacts, which need to be assessed to guarantee the sustainable management of water-resource systems. Understanding the allocation of water in catchments (or river basins) and its impacts in economic and hydrological dimensions is crucial in this context [7].

Hydrological and economic tools have been commonly used to model hydrological and socio-economic interactions in order to assess the impacts of certain policy measures in specific hydrological and climatic contexts. At the policy level, the use of integrated multi-disciplinary methods (e.g., hydrology, engineering, and economics) to support water decision-making has been promoted for the assessment and development of sustainable water-management strategies in integrated water-resource management (IWRM) [8,9]. One example is the paradigm shift represented by the EU Water Framework Directive (WFD) that imposes the use of economic science, including the use of scenarios in the characterization of water uses (Art 5) and the consideration of economic instruments in order to reach sustainability goals (Art 4 and Art 9) [10]. In line with this reasoning, hydro-economic models (HEMs) have been widely used by academics and policy-makers in recent decades.

This study aims to offer an updated review of the advances in hydro-economic modelling in the last decade, focusing on the assessment of water management in the context of a changing climate. Ever since the general reviews were published at the end of the first decade of the current century [11–15], significant advances have been achieved regarding both the assessment of the impacts of water-policy instruments (e.g., water markets, water banks, insurance instruments) at a river basin or catchment level, and regarding the consideration of CC implications in HEMs. In contrast to recent general reviews [16], this work focuses on recent developments of river-basin HEMs for the analysis of water policy instruments in the context of CC.

With this aim in mind, Section 2 offers a brief overview of HEMs and definitions, followed by a classification of alternative approaches used during the last decade in hydro-economic modelling to address water-resource management issues and CC at the river basin (or catchment) scale (Section 3). Subsequently, Section 4 discusses recent advances achieved regarding the assessment of water-policy instruments in the context of CC through hydro-economic modelling, while Section 5 centres the analysis on how uncertainty and risk factors of global climate change have been addressed in recent HEMs. Finally, Section 6 offers a brief discussion and some concluding remarks.

2. Overview and Definitions

HEMs arose from the combination of water-resource planning models with considerations of welfare economics in the 1950s [9]. In those years, Krutilla and Eckstein [17] considered the basin as "the natural scale for hydro-economic modelling", thereby challenging previous methods based on a sectoral division. Furthermore, this conceptual development clearly established that both quality and quantity of freshwater are affected by all water users, accepting that all uses are hydrologically connected at catchment scale. The work of Vaux and Howitt [18] constituted one of the first applications of HEMs at a regional scale for the assessment of water transfers in California. Subsequently, Booker and Young [19] extended the approach in order to account for all hydro-economic and socio-economic characteristics of the Colorado River basin.

In the last two decades, HEMs have incorporated an integrated analysis of impacts related to CC on water-resource systems, both spatially and temporally [16,20]. River basins in arid and semi-arid regions worldwide face major challenges in water scarcity, which will probably be aggravated by CC in the coming decades. In this context, HEMs play a major role for informing water policy and advancing sustainable use of water resources. One advantage of HEMs is the capability to capture the interrelationships between economic, hydrological, institutional, and environmental dimensions for a comprehensive assessment of the trade-offs among water-policy options [14]. Potential impacts of CC have largely been assessed using the approach developed by Hurd et al. [21], Hurd et al. [22], and Hurd and Harrod [23], based on the use of different climatic scenarios. At a river-basin scale, a first attempt was carried out in the upper Rio Grande basin, where Ward et al. [24] developed an HEM to assess the effects (i.e., socio-economic, hydrological and environmental) of sustained drought at the catchment level, which was further extended by Ward et al. [25] to include the protection of endangered species in the same basin. Finally, HEMs also address questions regarding the adequacy and sustainability of water-supply sources and infrastructures under changes in climatic conditions [9,26]. These models have recently shown significant capacity to identify strategies for the improvement of various sets of policy decisions, such as investments to improve irrigation efficiency, infrastructure design, and institutional reforms in a global change context [27]. Examples of recent work that addresses the potential impact of CC on water supplies include the works of Jeuland [28], Tilmant and Arjoon [29], and Amin et al. [30].

Throughout the advancement of research in the context of water resource management and linkages between socio-economic and physical aspects of hydrological systems, various terms have been used to describe the models applied. The bandwidth of terminology ranges from "hydrologic-economic" [31], "economic-hydrologic-agronomic" [32], "integrated economic-hydrologic" [33,34], and "holistic water resources-economics" [13,35], each highlighting the predominant factors and the weight of the hydrological vs. the economic model components, as well as the methods and the spatial and temporal scope of the analysis. Hereinafter, this review uses the term 'hydro-economic', as used in the latest reviews for models that combine hydrological and economic components to analyse water-resource systems [16,36].

While Bekchanov et al. [16] differentiate economy-wide models (referring to computable general equilibrium (CGEs) and input-output models) and network-based HEMs, our definition of HEMs focuses on the latter, as they are especially relevant for the analysis of water policy issues at a river-basin scale. They combine microeconomic theory and stochastic hydrological operation models and can be differentiated into simulation models and optimization models [16]. HEMs differ from economy-wide models, since the latter aim to widen the analysis to include the general and/or global economy (e.g., by considering inter-sectoral linkages and trade exchanges). An interesting review of these economy-wide models is presented by Dinar [37]. In contrast, HEMs focus on river basins or catchments and analyse specific water-management solutions at this scale. Similar to network-based models, river-basin HEMs use simulation and/or optimization methods within the wider concept of water-economy models and aim to integrate hydrologic and economic systems to provide appropriate policy (e.g., allocation, infrastructure) solutions at different spatial and temporal scales [16]. These models can be used to assess future scenarios in water-resource systems when external shocks (climate change, macroeconomic conditions, infrastructure, policy decisions, etc.) occur.

3. Classification of HEMs

Following Cai et al. [38], HEMs can be classified as either holistic or compartmental models. While compartmental models are constructed on separate modules (e.g., economic, hydrological) that use each other's input/output data, holistic models are designed to integrate all modelled aspects in a single consistent framework. In compartmental HEMs, feedback loops are generally needed, which require appropriate model interfaces between alternative compartments.

Furthermore, network-based HEMs can be differentiated into simulation models and optimization models [16]. Hydro-economic simulation models are employed to assess specific "*what if*" scenarios (such as climatic conditions) for certain management decisions. Simulation models are suitable for the exploration of precise and specific management policies and for the exploration of the ability of a quantitative approach to simulate the behaviour of certain variables. Moreover, simulation models can be applied both at smaller and larger water-resource scales to examine the effects of specific water-management strategies and behaviours at different management levels [39]. The main disadvantage of simulation HEMs arises from the problem to identify the best policy option under the various model scenarios that may potentially be considered.

In contrast to simulation models, hydro-economic optimization HEMs can help to identify "*what's best*" and assess alternative decisions and action sets within natural and human-made constraints, such as the availability of water resources and institutional and legal issues. Optimization techniques, such as linear and dynamic programming, largely focus on water-allocation optimization and profit maximization and are generally applied to assess water-allocation decisions subject to the maximization of water-use economic gains under certain environmental constraints, such as water availability. This approach has recently been extended to include the impacts of alternative water uses on water quality, catchment ecology, and non-market economic values [16].

There is no dominant modelling approach (simulation vs. optimization), since the management of extreme events (droughts and floods) must handle uncertainty and the likelihood of event occurrence while water policy analysis relies on the identification of optimality assessments. To address and counteract the limitations of each modelling strategy, simulation and optimization methods can be combined. Such "hybrid" models enable the results from optimization models to be tested and refined with simulated outcomes [40]. This approach has been extensively used in recent years. Both simulation and optimization models constitute constructive approaches for the implementation of IWRM alternatives to address socio-economic and legal-political objectives, thereby also facilitating the integration of stakeholder concerns and the implementation of adapting water-resource management to changes in climatic conditions. Meanwhile, integrated and sometimes dynamic hybrid HEMs are increasingly being applied in order to consider shifting conditions of greater complexity, especially those concerning potential CC impacts and scenarios [16]. Along similar lines, Herman et al. [41] argue that hybrid HEMs can be extremely helpful in exploring potential CC concerns by identifying vulnerabilities of water-resource systems and adaptation strategies.

An alternative approach is the inclusion of stochastic elements in optimization HEMs. This opens another line of differentiation between deterministic and stochastic modelling approaches. Most HEMs assume perfect foresight, but river basin managers cannot perfectly foresee water availability and have to deal with high risks [42]. Such risks can be accounted for by the use of a variety of possible future scenarios (hybrid models, see above) and/or by including stochastic risk components in the optimization problem [43].

As Hanemann [44] remarks, water resources are subject to challenges derived from institutional settings and property-right schemes and to the conflicting interests among multiple agents. While the hydrological component helps to reveal where the water is distributed to in physical terms, the economic component contributes by considering the net economic values of any such distribution. Therefore, the combination of economic modelling with hydrological processes provides a more realistic framework for the analysis of potential impacts of climate-related issues on the management of water resources at a catchment scale [12,45]. Assessments by HEMs can lead to useful findings to report water allocation and policy decisions, as well as other economic and performance results, such as water-use values, management and the construction of supply infrastructures, as well as the design of sectoral policies (e.g., agricultural policies) [14,46]. Along these lines, HEMs are often classified into hydrological management models (e.g., assessment of water-infrastructure design or management), and policy and allocation models that are mainly focused on the efficient management of water resources under certain spatial, hydrological, and climatic conditions. This work focuses on this latter type of HEM.

4. Recent Developments

Changes in water and environmental policies are generally catalysed by external factors, such as political, economic, and sectoral interests (e.g., agriculture, industrial), and extreme events that modify the prevailing water conditions [47]. Any model designed or implemented to support policy analysis should be realistic and sensitive to changes in critical variables (e.g., water availability, prices of agricultural products) and should allow for changes in the operational rules of the water infrastructure, climatic variables (e.g., water supply, temperature), characteristics of decision-makers (e.g., farm size, household size), and decisions on policy instruments, such as subsidies, taxation, input pricing (e.g., water, pesticides), quantitative limits (e.g., water abstraction, discharge limit, fertilizer use), and technology adoption (e.g., water efficiency use, energy mix). Since the work involved in covering all these topics is beyond the possible scope of analysis, most models are driven by sectors and focus on specific policy options (e.g., prioritize infrastructure investment decisions, water pricing). Among the different applications of HEMs, there are many examples with a main focus on water-quality issues (e.g., [48–52]), water-allocation strategies (e.g., [34,53–56]), water-policy instruments (e.g., [57–59]), and land-use planning policies (e.g., [60]), among other concerns. Furthermore, interest in the assessment of the impacts of and adaptation to CC of water-resource systems, and the consideration of the associated uncertainties and risks to various climatic scenarios in the application of HEMs have attracted increasing attention [14,43,61–68].

The recent bibliometric review by Bekchanov et al. [16] shows that the largest number of studies using HEMs in recent years have focused on the impact of climate on water-resource systems and the assessment of adaptation policies to decreasing water availability. Obviously, human activity and extreme events affect hydrological balance and both need to be considered in hydro-economic modelling assessments, otherwise modelled outcomes could lead to sub-optimal decisions and to an increase in risks for the viability of economic activities and the sustainability of water-dependent environments. The most recent HEMs take into account rising global warming, the increasing risks of extreme events (i.e., drought and floods), and their negative consequences in terms of economic losses, food security, and human health, among other factors, in order to identify promising adaptive measures and to provide accurate information for decision-makers [43,69]. Nevertheless, despite these efforts, there are still very few HEMs that address the potential impacts of CC on water-resource systems at a river basin scale, and even fewer studies consider the interlinkages of physical and economic extreme events effects in terms of the costs and also the benefits of potential adaptation actions [68].

To analyse water policies at basin scale, both qualitative and quantitative models have been applied. The process of prioritizing public policies for economic development implies the need for quantitative and qualitative models that support ex-ante policy evaluations as close as possible to the complexities of the real world. Hydrological models are used by engineers, water agencies, and for land-use planning and constitute a necessary tool for water and environment resource management. When the system under analysis integrates both a hydrological model and the socio-economic factors, it can be used for policy assessment and evaluation of water management decisions. Qualitative models have frequently been used to support policy making. One example is the Driver-Pressures-State-Impact-Response (DPSIR) framework [70], employed by EU institutions and other institutions such as OECD [71]. One evolution from this DPSIR framework is the systems thinking approach, for more advanced qualitative modelling at a basin level to support policy decisions. Mai et al. [72] used a systems-thinking approach to develop a conceptual model of a water-trading scheme in Australia. This is essentially a holistic approach that accounts for interrelations of a system's constituent parts, and it has been used in the construction of economic-environment scenarios.

Still in the field of microeconomics, with the support of mathematical programming techniques and database management, several relevant models take into consideration the fact that water policy is closely related to land-use policy, since agriculture is the main user in many regions of the world [73]. This is especially true for arid and temperate regions that are on course towards basin closure or have already reached a mature economy state where demand surpasses the available supply. Therefore, land-use agricultural models that include water-management decisions are frequently seen [43,69,73].

The EU normative that promotes ecosystem-based thinking as a way to influence policy- and decision-making should account for the behaviour of natural resources. In this line, the Blueprint to Safeguard Europe's Water [74] aims to inform the EU water policy through the assessment of both quantitative and qualitative aspects of water resources, thus taking into account climatic and environmental issues and offering water balance assessments at catchment (or basin) scales [74]. A major part of this water balance involves accounting for water removed from rivers or aquifers by different sectoral needs, which are significantly impacted by CC. Mubareka et al. [75] used the CAPRI (Common Agricultural Policy Regionalised Impact) model for EU agriculture and integrated a water module that had previously been used for scenario building to analyse the impact of Common Agricultural Policy (CAP) measures under different scenarios. Blanco et al. [73] assessed the role of climate change as a driver of the agri-food systems and include agricultural water demand. This model uses microeconomic/macroeconomic integration since farmers' decisions affect local and global market outcomes, and therefore also affect local and world prices. CAPRI is mainly a land-use and farm economic model for agricultural policy support. Other models integrating micro- and macro-components are more detailed regarding hydrological impacts. Parrado et al. [76] included two-way feedbacks, decentralized irrigators, and a regionally-calibrated CGE model to assess interlinkages.

Engineering-based hydrological models, which were originally developed for water management and infrastructure operation, usually include an economic module of cost minimization, profit maximization, or management of supply failure. These models are usually labelled as operational models. As part of the evolution of the operational models, several economic modules are integrated that are part of engineering processes and respect the allocation of water rights and operational rules to simulate marginal changes in water supply or demand and to evaluate changes in the value of the production functions. The AQUATOOL model for Spain [77] and CALVIN model for California [41] constitute good examples of hydrological models that integrate such economic considerations. These models seldomly integrate macroeconomic feedbacks, and the level of representation of sectors becomes increasingly complex in order to be as close as possible to real decisions including some behavioural models that are more accurate than neoclassical profit-maximizing assumptions. These models operate at the basin or sub-basin scale as management units. The time scale is usually monthly in order to include water storage and seasonality of demand. Uncertainty in water resources, including the occurrence of extreme events, such as floods and droughts, is usually included by integrating available information regarding past climate observations or future projections [68]. An example of an HEM focusing on infrastructure valuation for energy and irrigation is the model WHAT-IF [78], where the objective is to maximize economic welfare expressed in terms of the sum of consumer and producer surplus subject to environmental, physical, and institutional constraints.

Since water and energy systems are interlinked [7], the number of models addressing the food-energy-water (FEW) nexus is growing. Brouwer et al. [79] reviewed six key models employed to support policy-making institutions (European Commission, OECD, and the World Bank). Many of these models give priority to the evaluation of energy (hydropower) and irrigation, as does WHAT-IF model [78,80]. Recently, certain models, such as CLEWs (Climate, Land-use, Energy and Water strategies) [81,82], have included the FEW nexus. At this point, we believe that CC mitigation and the integration of food production, irrigation, and energy use are critical and should be considered as a set when designing agricultural or water policies. An example of this is given by the promotion of solar-based pumping for irrigation, which may have advantages for energy policies but may also exert potentially negative impacts on the environment caused by excessive resource abstraction [83].

Another major development in recent years has been the integration of potential CC impacts in hydro-economic analyses. Jeuland [28] emphasized the importance of integrating both the potential CC effects on physical water availability as well as the economic implications that arise due to CC. Many analyses addressing scenarios of CC in HEMs have mostly merely focused on the physical aspects and have ignored not only the economic uncertainty and risk components but also crucial feedbacks between the economic side of water demand and physical water supply. The successful inclusion of feedback effects and model linking to account for the full range of physical and socio-economic global change effects is a major challenge in hydro-economic modelling.

A primary element of the study of the potential effects of CC involves the identification of system vulnerabilities and the measurement of system performance under possible projected climate scenarios. The IPCC defines vulnerability as "a function of the character, magnitude, and rate of climate variation (climate hazard) to which a system is exposed, and of non-climatic characteristics of the system, including its sensitivity, and its coping and adaptive capacity" [84]. Vulnerabilities associated with CC and extreme hydrologic events (drought and floods) determine constraints for an adequate performance of water-resource systems and affect both demand and supply. The identification of such vulnerabilities is key to the development of successful climate-adaptation strategies. On the demand side, recent HEMs have focused on the assessment of water allocation among alternative uses based on the economic value of scarce water resources and the interactions between alternative stakeholders that share surface and groundwater resources in an increasingly complex climatic and hydrologic context (e.g., [20,27,42,82,85]). Under this approach, most studies aim to inform management decisions under conditions of water scarcity and increasing vulnerabilities of water-resource systems likely due to CC.

Table 1 summarizes recent studies that have applied HEMs to the management of water-resource systems that take into account the vulnerabilities and uncertainties associated with CC (and extreme events, such as drought and floods). This summary also offers general information regarding the model type, methods, case study, consideration of surface water (SW) and/or groundwater (GW), and sectors implied (e.g., irrigation, urban, hydro-power, environment).

Table 1. Review of recent applications of HEMs that incorporate potential CC effects.

Study	Models & Method	Model Type	Main Focus	Case Study (Country)	Water Type	Sectors
Amin et al. [30]	Water evaluation and planning (WEAP) model	Simulation, compartment	Water supply & climate change	Upper Indus river basin (RB) (Pakistan)	SW & GW	All sectors
Souza da Silva and Alcoforado de Moraes [42]	Soil and water integrated model (SWIM) + Model of agricultural production and its impact on the environment (MAgPIE) /General algebraic modeling system (GAMS) + Positive mathematical programming (PMP) models	Optimisation, holistic.	Trade-offs between uses & climate change	Sao Francisco RB (Brasil)	SW	Hydropower, urban, irrigation, & environment
Essenfelder et al. [86]	Positive multi-attribute programming (PMAUP) + Soil and water assessment tool (SWAT) models	Hybrid, holistic	Climate change & adaptation strategies	Mundo RB (Spain)	SW & GW	Irrigation
Herman et al. [41]	CALVIN model	Optimisation, holistic	Water-supply & climate change	California (USA)	SW & GW	All sectors
Escriva-Bou et al. [68]	AQUATOOL + (Simulation model for watershed management) SIMGES	Hybrid, holistic	Climate change & adaptation strategies	Jucar RB (Spain)	SW & GW	All sectors
Ruperez-Moreno et al. [85]	HEM for Segura RB	Optimisation, compartment	GW management & climate change	Segura RB (Spain)	GW	Irrigation & environment
Kahil et al. [20]	Modular finite-difference flow model (MODFLOW) + GAMS + PMP	Hybrid, holistic	Trade-offs among water policies & climate change	Jucar RB (Spain)	SW & GW	Irrigation, urban (large cities), & environment
Esteve et al. [67]	WEAP-MABIA modelling framework + PMP	Optimisation	Climate change & adaptation strategies	Middle Guadiana basin (Spain)	SW & GW	Irrigation
Kahil et al. [77]	AQUATOOL + Jucar RB optimization model	Hybrid, compartment	Water scarcity, droughts & climate change adaptation	Jucar RB (Spain)	SW & GW	Irrigation, urban, & environment
Kreins et al. [87]	Water balance model mGROWA + climate model WETTREG	Integrated model framework	Climate-change impacts on irrigation & GW management	North Rhine-Westphalia (Germany)	GW	Irrigation
D'Agostino et al. [43]	Non-linear optimization model + hydrological GIS-based model + CLIMAWARE	Optimisation	Climate-change effects on water balance	Apulia (Italy)	SW & GW	Irrigation
Tilmant et al. [29]	Stochastic Dual Dynamic Programming (SDDP) model	Optimisation, compartment	Water-supply & climate change	Euphrates RB (Turkey, Siria)	SW	Hydropower & irrigation

Table 1. *Cont.*

Study	Models & Method	Model Type	Main Focus	Case Study (Country)	Water Type	Sectors
Yang et al. [65,69]	Indus Basin Model Revised-Multiyear (IBMR-MY)	Optimisation, compartment	Allocation strategies & climate-change adaptation	Indus RB (Pakistan)	SW & GW	Irrigation
Hurd & Coonrod [27]	Water balance (WATBAL) model + circulation models (temperature and precipitation), GAMS	Hybrid, holistic	Trade-offs between uses & climate-change adaptation	Upper Rio Grande (USA)	SW & GW	All sectors
Harou et al. [88]	CALVIN model	Optimisation, holistic	Trade-offs between uses, drought & climate change	California water system (USA)	SW & GW	All sectors
Jeuland [28]	Standard water resources planning model + Montecarlo methods	Simulation, compartment	Water-supply & climate change	Nile RB (Egypt)	SW	Hydropower & irrigation
Varela-Ortega et al. [59]	WEAP model	Optimisation	Water and agricultural policies & climate change	Upper Guadiana basin (Spain)	SW & GW	Irrigation
Reynaud and Leenhardt [89]	Model for water resources management (MoGIRE)	Optimisation holistic	Integrated water management & climate change	Neste basin (France)	SW	Irrigation, urban & environment
Tilmant and Kelman [90]	Stochastic Dual Dynamic Programming (SDDP) model	Optimisation compartment	Water-supply & climate change	Eurphrates and Tigirs rivers (Turkey)	SW	Hydropower & irrigation
Tanaka et al. [61]	CALVIN model	Optimisation holistic	Climate change & adaptation strategies	California water system (USA)	SW & GW	Irrigation & urban

Source: Authors' Own.

It is worth noting that many studies only include one selected CC scenario, without considering the range of CC risks and uncertainties [42,67,87]. For example, Kreins et al. [87] only considered one CC scenario (SRES A1B1 scenario), even though they aimed to assess CC impacts on GW resources in North Rhine Westphalia (Germany). Therefore, they failed to explicitly address CC uncertainty and only include one possible future temperature and precipitation trajectory. In order to address the uncertainty related to CC and risks in the management of water resources, several global and regional CC and GHG-emission scenarios should be included [87]. Souza da Silva and Alcoforado de Moraes [42] used a basin-wide hydro-economic optimization model to analyse trade-offs regarding water-management decisions in the São Francisco River Basin in Brazil. They constructed various operating-rule scenarios under certain institutional constraints and compared the outcomes of shadow prices of reservoir outflow and associated costs and benefits. They did so under a baseline scenario without CC and compared their results to a scenario under CC following the IPCC SRES A2 climate-change scenario [84]. They used this HEM to evaluate the economic effects of different management options ("operating rules"), environmental, technical, and institutional constraints, as well as land-use change and CC, and identified optimal water allocation between various water users.

5. The Challenge of Uncertainty

In order to provide informed policy advice and to assess the real costs, benefits, and associated risks of water infrastructure investment projects under future CC, it is of prime importance to take the uncertainties associated with CC into account. This is the case in recent HEMs, as summarized in Table 2. For example, D'Agostino et al. [43] used a sensitivity analysis of the major water balance components for their hydro-economic analysis of water use in the agricultural sector of Apulia (Italy). Their results revealed that climatic conditions, soil type, and cropping patterns exerted a major impact on the outcome of the model. The variance of the upper and lower bounds of irrigation water requirements (with a lower bound of 39% and an upper bound of 103%), groundwater recharge (40–53%), and surface runoff (46–59%) show that irrigation water requirements are especially prone to uncertainties of climatic conditions [43]. Ignoring this variance and solely providing point estimates would bias the water-planning decisions.

The consideration of potential CC effects on HEMs introduces many forms of uncertainties into these models. On the one hand, there is data and input-parameter uncertainty: (a) regarding physical input parameters (e.g., precipitation, runoff, among others); and (b) with respect to economic inputs (e.g., water demand, water prices) [28,42,43,68,91,92]. On the other hand, there is major model uncertainty: first, inherent model uncertainty of climate models [84,93,94]; second, model chain uncertainty from deriving information from global to regional data and from regional to spatially more explicit climate data [91]; and third, there are biases involved when using upscaling and downscaling methods [42,68].

Due to the conjunction of hydrological and economic components, HEMs are prone to data and input-parameter uncertainty from both the physical side of water availability as well as the socio-economic side of data demand and its complex interlinkages [95]. Even in the absence of relevant CC effects, the physical availability of water itself is highly uncertain in nature due to short-term weather variations and upstream water extractions by other water users. There is temporal (seasonal, annual, long-term) variation as well as spatial variability in water supply. Similarly, crop water requirements are highly uncertain [92,96]. These uncertainties are amplified in the presence of CC. Changes in climatic conditions and precipitation induce biophysical and hydrological uncertainties as well as socio-economic risks [28,42,43]. Through changes in rainfall patterns, glacier melt, recharge rates, runoff flows, extreme events (floods, droughts, storms, heat waves), and sea-level rise, CC affects the availability of usable freshwater [28,68]. On the demand side, household, wastewater treatment, industrial, and agricultural freshwater demand are affected by changes in ocean and surface temperatures and precipitation patterns. Changes in plant growth, crop water requirements, and evapotranspiration all influence irrigation water demand [28]. Industrial water demand might increase due to greater cooling requirements and due to the complex links of energy prices that might increase demand for hydropower. Moreover, environmental water needs might increase due to potential CC impacts (e.g., due to saltwater intrusion in costal ecosystems associated with sea-level rise [84]). There are feedbacks between water-management decisions, socio-economic effects, and water availability that further increase uncertainties and risks in HEMs [43,91].

Input parameter and data uncertainty can be addressed by: (a) various scenarios combined with a sensitivity analysis in the case of simulation-HEMs, or (b) stochastic programming, that is, through the introduction of a stochastic component in the optimization of the model [43]. Most optimization-HEMs are deterministic in nature. Deterministic models fail to account for uncertainties in the variables and parameters used [97]. In order to account for such uncertainties, a stochastic component can be included in the optimization model. Input-parameter uncertainties can be included in the objective function by including risks in crop prices, yields, incomes, and resource-availability constraints [92]. In this setup, expected profits or expected utility rather than deterministic profit/utility/gross margin functions are maximized. Statistical modelling of input-parameter uncertainty can, for instance, be achieved by “stochastic programming” or “discrete stochastic programming”. The latter includes more than one decision stage and a revision of the decision taken by the farmer. Graveline et al. [97] compared the results from a deterministic approach by analysing three different global-change scenarios with respect to climatic conditions, the economic environment, and the regulatory environment with a Monte Carlo approach using 200 random selections under these three scenarios. They showed that the discrete solution of the deterministic model is prone to false conclusions, since it fails to account for uncertainty. In order to provide informed policy advice, it is important to account for uncertainties of input parameters in mathematical programming models.

In order to account for CC impact uncertainties, different emission and CC scenarios can be applied to HEMs. To this end, local HEMs need to be combined with global or regional climate models.

However, feeding HEMs with output data generated by climate models amplifies model uncertainties in HEMs [91,98,99]. Both global hydrological models (GHMs) and global climate models (GCMs) have inherent uncertainties that are translated into HEMs and may be even. Irrigation water demand varies substantially across different global hydrological models (GHMs) and global climate models (GCMs). According to Wada et al. [99], uncertainties from GHMs exceeds GCM uncertainty along the projection period until 2100. While GHMs show constantly significant uncertainty throughout the whole century, uncertainty in GCMs increases along the projection period). According to Döll [100], there is more variation in the outcomes of the models arising from differences between the various climate models applied compared to the differences between the various emission scenarios. Introducing potential CC effects in catchment-based hydrological or hydro-economic models requires the downscaling of results from regional climate models that in turn derive their outcomes from global climate models. This introduces additional uncertainty in HEMs [93]. Sophisticated methods are available to conduct downscaling with bias-correction methods of global to regional information regarding land use and climate change [65,101]. In order to meet the demands for local HEMs, these regional data need to be downscaled even further to obtain climate information at a basin or catchment scale. This process involves uncertainties and biases that are often ignored in HEMs.

In order to address model uncertainty, model chain uncertainty, and upscaling/downscaling biases, various global models can be applied as robustness checks of the analysis [99]. Previous research shows that model selection is crucial when analysing CC impacts in the context of water resources. It is recommended to employ several hydrological models and various emission or climate scenarios [98,99,102]. Wada et al. [99] suggested a multi-model approach to address uncertainties arising from model uncertainty and CC uncertainty in their analysis of irrigation water demand in order to provide robust modelling results.

The majority of HEMs addressing CC risks and uncertainties apply simulation models. Escriva-Bou et al. [68] selected six regional climate models that showed the best-fitting results when compared to historical precipitation and temperature data in the basin analysed (Jucar River basin, Spain). Graveline et al. [66] constructed one CC scenario by downscaling precipitation, temperature, and climate data from regional climate models (ECHAM4/OPYC3 [103], ENSEMBLES EU-project [104], Rossby Centre regional Atmosphere-Ocean Project RCAO [105], and PRUDENCE simulations [106]) and combined them with two catchment-specific agricultural management scenarios (water-storage capacity and irrigated land increase; modernization of irrigation technology) to address the effects of climate and socio-economic changes on water resources in the Gallego catchment area (Spain). D'Agostino et al. [43] used an integrated HEM for the case study area of Apulia in Italy to assess impacts of CC on the water balance and agricultural water use. They explicitly accounted for uncertainty by considering different CC scenarios and by conducting a nominal-range sensitivity analysis. Further to the commonly used A1B SRES emission scenario, four additional CC scenarios were selected. Sensitivity analyses were employed to determine the contribution of single-input parameters to variations in the simulation model output [43]. This enabled the response of input parameters to be assessed that are likely to suffer from uncertainty. Sensitivity analyses are commonly used in HEMs to gain information on outcomes of groundwater recharge, runoff, or crop evaporation under changing rainfall and temperatures [107].

Table 2. Consideration of CC-related uncertainties and risks in HEMs.

Type of Uncertainty	Treatment of Uncertainty	Examples
Input-parameter uncertainty (physical)	Stochastic programming	[28,29,90]
	Sensitivity analysis	[41,43]
Input-parameter uncertainty (economic)	Stochastic programming	[28,43,59,67,89,97]
	Sensitivity analysis	[41,43,77]
Climate uncertainty	Several climate-change scenarios	[20,30,41,43,61,65,68,69,77,86,88,97,106]
Model (chain) uncertainty (Upscaling/downscaling)	Use of different (global/regional) climate models	[27,41,43,65,66,68]

Source: Authors' Own.

Other HEMs apply stochastic methods to their optimization model. D'Agostino et al. [43] included stochastic components in their optimization model. The non-linear stochastic economic component of the HEM that maximizes farmers' utilities takes uncertainties with respect to prices and yields into account. Jeuland [28] used the concept of hydro-economics as an investment planning framework and took the interrelationships between CC and water-resource systems into account. These last two references included both physical aspects of CC (changes in runoff, net evaporation, water demand, and flood and drought risks) as well as economic uncertainties (e.g., real value and productivity of water-system-related goods and services). The innovation of this approach involves extending a hydrological water-resource planning model to include economic uncertainty. Additionally, Jeuland [28] accounted for uncertainties by using a stochastic streamflow generator, a hydrological simulation model, and an economic appraisal model. Regarding CC, the author applied a historical scenario and a scenario based on the SRES A2 emissions scenario presented in the IPCC report [108]. The economic appraisal model calculates the net present value (NPV) of hydrologic projects under a Monte Carlo simulation and considers various possible physical and economic states. Reynaud and Leenhardt [89] took economic risk into account by introducing a probabilistic component in the microeconomic production model and represented each farmer's behaviour in their integrated water-management framework, thereby representing agricultural, urban, and environmental water demand in the case of the river Neste (France). This model includes climate and crop price variation and farmers' risk preferences and influences farmers' choices regarding land use, sowing dates, and water use. Alternatively, Graveline et al. [97] conducted Monte Carlo simulations in order to account for input-parameter uncertainty in their farm-scale model applied to two regions in France, and Varela-Ortega et al. [59] considered uncertainties via stochastic programming methods in the economic model, which is combined with a hydrological model to form an HEM.

6. Discussion and Concluding Remarks

Despite the recent developments in the use of river-basin network-based HEMs to assess water policy in a CC context, several remaining challenges can be identified. One important decision in the context of hydro-economic modelling is the spatial scale of analysis to be used. It is of crucial importance since it may introduce further uncertainties into the model due to aggregation bias or upscaling/downscaling procedures [92]. Clearly, spatial scale depends on the research question or policy evaluation to be addressed. Although the farm scale may be useful to analyse farm decisions and impacts on different farms, regional or catchment models are optimal to determine the social optimal allocation of water resources. However, this scale can only be applied in a relatively homogenous region [109]. Additionally, the models may suffer from aggregation bias. This is especially relevant for water resources, since water availability and use are often heterogeneous within a region [92]. The river basin (or catchment) scale has been acknowledged as the appropriate scale of analysis to address CC challenges in water-resource management [110], since modelling at this scale can provide essential information for policy makers in their decisions regarding the allocation of resources [33].

Furthermore, non-provision ecosystem services, such as environmental and other in-stream water uses, become increasingly important when economies develop, whereby the basin scale presents the most suitable unit of analysis. In contrast, the use of economy-wide models that include water use are inadequate for water-policy decisions since the lack of hydrological details (e.g., water resources, water abstraction, return flows, temporal evolution) and the level of analysis (e.g., country/region) makes them unfit for specific water-policy evaluation. These economy-wide models fail to recognize a critical variable regarding water use: return flows. These flows are crucial for water analyses since most of the water in many sectors (energy, urban, industry) returns to the system (with lower quality but almost in the same quantity, and usually from a different location as that of abstraction). The global average agricultural return flows are close to 40% (i.e., only 60% is evaporated or "lost from the basin") [47]. Modelling water policy requires this information to be taken into account in order to make a realistic and useful model.

At the same time, micro-macro linking becomes increasingly important in certain modelling contexts. Most applications of HEMs consist of microeconomic analytical tools that include water as the resource under analysis within a neoclassical optimization framework to evaluate specific policy measures. Among these applications, Berbel et al. [111] focused on the hydrologic and economic impacts of water-use efficiency upgrading, and Xie and Zilberman [112] evaluated water-supply increases vs. water-use efficiency policies. These models tend to focus on specific temporal and spatial contexts while ignoring larger scales such as the river basin (or catchment) and climatic variability. Regarding the use of models for policy evaluation, generally, HEMs are built from detailed hydrology models and integrate relevant sectors, mainly irrigation and energy (cooling and hydropower). However, the regional macroeconomic effects of a range of water allocation and investment decisions are generally not considered in most models. They should be integrated as micro-macro feedback loops (e.g., less irrigation, reduced output, multiplier effect, higher prices, consumer impact, and welfare effects). Hitherto, such analyses have seldom been carried out in the literature. Regarding the models of the microeconomic sector, the use of mainstream neoclassical economics, which relies on the optimizing behaviour of agents to determine microeconomic decisions and to link these to macroeconomic decisions, should integrate the insights of behavioural economics in order to improve the usefulness of the model and to improve the predictive capacity of models and the effectiveness of policies. Along these lines, and in contrast to most applications of HEMs reviewed in previous studies ([12,14,16], among others), the use of HEMs at river basin scale should take into account three basic dimensions (or components): hydrological, microeconomic (bottom-up), and macroeconomic (top-down). CC would enter the HEM as an element that influences water and socio-economic systems and incorporates variability and uncertainty into the modelling assessment.

In the context of potential CC impacts, this study highlighted the range of uncertainties (input-parameter uncertainty; scenario uncertainty; model chain uncertainty) that have to be addressed by the models [28,42,43,68,91,92]. Climate-change and global-change (i.e., bio-physical, regulatory, economic conditions) uncertainties can be included by employing alternative possible future scenarios regarding emissions, agricultural policies, prices, and resource constraints [43,92]. More specifically, in order to account for CC uncertainties, optimization models or descriptive models of agent behaviour can be complemented with simulation methods by including diverse scenarios representing different states of certain aspects (water availability, temperature, associated costs and benefits, environmental and economic circumstances, etc.) [28]. Alternatively, a variety of climate, environmental, socio-economic, and market conditions can be included by randomized statistical methods to directly include risk in the optimization model. In our opinion, the embedded uncertainty that is essential to any climate model should be managed inside the model by simulating various climate scenarios. For instance, such models should include several GHG-emission scenarios in order to account for the uncertainty related to future CC; they should take several global climate models into account for robustness checks and/or include stochastic components for both physical and economic input parameters. Furthermore, future economic growth should be considered in HEMs since the demand for food and energy substantially modifies the demand and supply of water. To summarize, CC uncertainties can be addressed by (a) including different CC scenarios in simulation HEMs or (b) incorporating stochastic components in optimization HEMs. Arguably, the recently more commonly applied hybrid approaches combining simulation and optimization network-based HEMs may be especially well suited to analyse water policies under CC at a river-basin scale.

This paper has reviewed the literature to categorize HEMs used for water-policy evaluation including the integration of CC impacts. This review updates previous efforts to describe the available approaches towards issues of supporting water allocation, infrastructure investment, and policy options. Although in recent years, several HEMs have started to take both bio-physical as well as economic factors and uncertainties and their feedback links into account, significant drawbacks and limitations still persist when they account for uncertainties and risks associated with CC. Thus, further research is needed to overcome these limitations.

To sum up, our main conclusion regarding CC uncertainties is that modellers are striving to introduce certain climatic scenarios. In the past, most of the HEMs that address CC focused on the physical impacts of changing climatic conditions while ignoring economic feedback or assuming fixed parameters for economic factors that are crucial for water management and investment decisions. This generally leads to errors in the valuation of costs and benefits of hydrological projects, especially in terms of socio-economic effects, such as the roles of agricultural adaptation, degradation, and migration, which cannot be addressed under such a setup. Current challenges in the realm of hydro-economic modelling include the representation of the food-energy-water nexus, the successful representation of micro-macro linkages and feedback loops between the socio-economic model components and the physical side, and the treatment of CC uncertainties and risks in the analysis.

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A Simplified Hydro-Economic Model of Guadalquivir River Basin for Analysis of Water-Pricing Scenarios

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Abstract: This study describes an economic model in the Guadalquivir river basin (Southern Spain) that considers inter-sectoral and hydrological effects of changes in water use as a response to various water-pricing policy scenarios. The main economic variables include water use, gross regional product, return flows in the river basin, and employment at sectoral and basin levels. The response of the different sectors to water pricing and of the sectoral productivity is derived from official data. The background of the model is based on previous research for the implementation of the UN System of Environmental-Economic Accounts and on the application of this framework to the Guadalquivir basin. Results based on the elicited curves illustrate that the structure of the demand function for irrigated agriculture passes from inelastic to elastic sections, while the function corresponding to the remaining economic sectors shows a continuous decreasing function with minor change in the elasticity structure of the curve. Results show that the impact of extreme measures of water pricing reduces water abstraction by up to 42% vs. the baseline scenario, with an economic reduction in regional Gross Domestic Product (GDP) of 1%.

Keywords: water pricing; water management; water policy; water-use efficiency; economic model; inter-sectoral; river basin

1. Introduction

Water scarcity and increasing inter-sectoral competition for available water resources exacerbate the need for an efficient and sustainable allocation of water. In this context, water-pricing policies have been considered as a suitable economic instrument to guarantee the efficient management of the resource and to deal with growing socio-economic pressure. A large body of literature has explored the effectiveness of water-pricing policies in managing demand in alternative sectors (households, industry, agriculture, etc.) and in achieving certain conservation goals (see, for example, [1–3]). Most water economists argue that price-based approaches towards promoting a more efficient use of water resources (especially in those locations suffering from water scarcity) and/or towards achieving conservation goals are more cost-effective than non-price-based approaches [4]. However, pricing reforms explicitly designed for these purposes are rarely observed. The work of [2] contains several case studies of water-pricing reforms over agricultural, industrial, and residential sectors, and arrives at the conclusion that certain political economy factors (such as the reason for the reforms, the interest and the parties involved, the existing institutions, and the power systems) prevent the implementation of theoretically efficient pricing reforms.

At European Union (EU) level, the Water Framework Directive (WFD) [5] requires EU Member States to implement economic instruments in order to manage water resources and to achieve a good environmental and chemical status of surface and groundwater bodies. Specifically, the Directive highlights the importance of estimating the economic value of water uses, the cost of the associated water services, and how much of that cost is recovered from users, and encourages the use of water pricing as a tool to achieve an efficient use of water. Nevertheless, little advance has been made in this direction. According to the Commission's Compliance Report [6] one of the main deficiencies in the WFD implementation involves the economic assessment of pricing measures and cost-recovery issues. Specifically, this report highlights the lack of methods for the calculation of costs (including environmental and resource costs) and benefits (including ecosystem services). Without these methods, neither will it be possible to ensure the implementation of effective pricing policies nor will disproportionate and inadequate measures be prevented.

Moreover, the WFD states that the level of cost recovery of water services should be analysed for certain water uses (including that of households, industry, and agriculture) and the characterization of water uses should refer to the basin as the level of management (Art. 5). Thus, the impacts of water-pricing should be both on a river basin scale and multi-sectoral. Finding ways to achieve positive economic outcomes in the management of water resources requires the aid of modelling tools to analyse the impact of alternative policy scenarios [7]. Following these recommendations, our model analyses not only the potential impacts of water-pricing policies (in various scenarios) on inter-sectoral water use and consumption, but also the effectiveness of these policies on the re-allocation of water between alternative uses within the river basin.

To this end, this study focuses on a strict economic point of view, since the main concept in order to determine water re-allocation among alternative uses is the economic concept of 'value'. The economic value of a given level of water consumption is driven by the benefit derived from its use. Water value changes with the quantity and type of use [8], and therefore monetizing water use enables a comparison to be made between uses and introduces clarity to the economic implications of water-management-related decisions. In a mature water economy [9], when demand exceeds supply, then another relevant concept is that of 'scarcity'. Water should be managed and allocated efficiently, that is, to maximize the value it provides to society. Under conditions of water scarcity, an economic focus, similar to that proposed in this study, helps identify efficient water allocations and reduce 'wasteful' practices. Additionally, the analysis of sectoral water demand and of its associated economic values of water facilitates the assessment of the effectiveness of public policies (i.e., water pricing), and identifies the trade-offs between resource uses.

There are numerous methods in the scientific literature for the assessment of the impact of re-allocation of water resources as response to economic policy measures, such as water pricing (see [10,11], among others). Nevertheless, studies have hitherto usually represented small spatial areas and/or addressed specific uses [12]. To the best of our knowledge, there are no studies available that analyse the effects of water-pricing policies on water use and consumption from a multi-sector approach and on a river basin scale where available water resources are depleted. This study aims to help fill this gap.

The proposed methodology simulates changes in water use for all relevant sectors in a river basin as the result of policy decisions regarding water-price measures. Price increases have been implemented by simulating various scenarios: baseline (current situation), financial and environmental cost-recovery scenarios, and two scenarios with major increases water costs. In order to test its applicability in a real context, the proposed methodology is applied to a specific case study: that of the Guadalquivir River Basin (GRB). The model requires a more detailed analysis of the irrigated sector, which is the greatest sector of consumption of water in the basin. The remaining economic sectors are taken into account via an estimation of water demand and economic productivity.

2. Materials and Methods

The Guadalquivir River Basin (GRB) contains 25% of Spain's irrigated land and it is the longest of the southern rivers (657 km); it can thus be considered one of the most important river basins in Spain. It covers an area of 57,679 km^2 and contains a population of 4.3 million. The basin has a Mediterranean climate with a heterogeneous distribution of precipitation. The annual average temperature is 16.8 °C, and the annual average precipitation is 573 mm, with a range between 260 mm and 983 mm (standard deviation of 161 mm). The main land uses in the basin are forestry (49.1%), agriculture (47.2%), urban areas (1.9%), and wetlands (1.8%) [13] (Figure 1).

Figure 1. Guadalquivir River Basin District. (Source: Guadalquivir River Basin Authority (GRBA)).

The GRB is considered a mature closed basin where most of the water resources are already allocated across various uses (agricultural and non-agricultural) and there are growing pressures for new activities to use 'additional' resources such as reclaimed water and new reservoirs. The key factor influencing this situation is the agricultural sector, which is the largest user of water, with irrigated agriculture accounting for approximately 88% of total freshwater withdrawals in the basin. Due to its high irrigation efficiency (as a result of an intense modernisation of irrigation over recent decades), irrigated agriculture is competitive but still yields lower returns in comparison with other uses (industry, tourism, urban areas) in the basin. As water becomes scarcer, society turns to agriculture as a potential source of water, in the sense that this is the sector of major consumption and therefore efficiency of the use of water in the agricultural sector directly affects the availability of the resource.

The proposed methodology for the economic model estimates sector-specific demand curves because water demand may change with location (e.g., up-flow and down-flow agriculture) and type of water use (e.g., urban, industrial, agricultural). Therefore, the primary aim here is to assess the competing demands between different uses on a river basin scale. Additionally, the analysis will apply an economic approach to the assessment of the effects derived from alternative water-pricing scenarios where water demands constrain total use of the available resource within a one-period analysis, and hence it has a static nature. The methodology presented in this study reveals a deterministic approach since it considers a single-set of fixed boundary conditions (e.g., hydrological conditions) and parameters (e.g., constant price-elasticity of water demand). Therefore, no stochastic-determined variables are considered in the model.

Economic sectors are classified according the importance and the water-use typology. The proposed sectors of the demand for water services in the basin are:

(1) Agriculture

- (1a) Rainfed agriculture
- (1b) Irrigated agriculture

(1c) Livestock

(2) Households
(3) Industry
(4) Services
(5) Recreation
(6) Energy

The valuation of water depends on whether the resource is considered an intermediate or a final commodity [14]. Water demand as an input to a production process (e.g., irrigated agriculture) can be derived upon the isolation of the marginal contribution of water to the total output value, and therefore a deductive estimation approach is required. Deductive techniques usually employ mathematical programming, although general equilibrium models and residual value methods also fall within this category. When water is a final consumption commodity (e.g., urban demand), inductive valuation techniques based on the econometric or statistical analysis of observed data to estimate price-response may be more appropriate. In Guadalquivir, as explained in greater detail below, either type of analytical approach is used, depending on the sector analysed. Regarding the agricultural sector, a deductive value methodology has been considered as more appropriate in order to assess crop and location differences across the GRB. Regarding the remaining economic sectors, a valuation based on estimated price-elasticities of water demand enable us to obtain water-use demand curves relative to changes in water pricing.

Therefore, the methodology used in this paper is organised in the following three phases:

2.1. Baseline Definition: An Appropriate Characterisation of the Economic Sectors in the Basin

Various sources have been used either for the observed original data or for the estimation of non-observed variables when necessary. The baseline scenario (Table 1) has been defined by employing the gross domestic product and employment by sector statistics from the Statistical National Institute, and the sectoral water use and prices from the Hydrological Plan by the Water Agency [13]. Global water abstractions in the GRB are estimated at 3614 Hm^3 in 2012, where irrigated agriculture constitutes the greatest sector of consumption with 88% of the total water abstracted. Economic activities in the GRB generated around €66.1 $\times 10^9$ in terms of GDP in 2012, which is equivalent to 7% of Spanish GDP. Over 73% of GDP in the GRB is concentrated in the service sector. Industrial activities amount to ≈18% of GDP, agricultural production ≈7%, and energy production ≈1%.

Table 1. Characterisation of the economic sectors in the basin. Guadalquivir 2012.

Sectors	Water Used (10^6 m^3)	GDP (10^6 EUR)	Employment (10^3 Person)	Price (EUR/m^3)
Rainfed Agriculture	-	1407	43	-
Irrigated Agriculture	3183.19	2585	79	0.060
Livestock	18.63	733	22	0.084
Households	261.00	-	-	1.900
Industry (non-energy)	68.00	12,175	228	1.112
Services	63.00	48,581	908	1.900
Recreation	1.00	10	0	0.025
Energy	19.00	626	12	0.049
Total	3613.82	66,117	1249	-

Source: Authors' own based on Statistical National Institute and [13].

2.2. Estimation of Demand Curves with Respect to Water-Price Changes for the Various Economic Sectors

2.2.1. Irrigated Agriculture Sector

The irrigation sector has been modelled by dividing the basin into two main areas (upper and lower basin) and by simulating demand curves in the current baseline scenario per crop area given the data available. Table 2 shows the characterisation of the irrigated agriculture sector (upper and lower areas) in the GRB in 2012. The upper area of the GRB is characterised by a more diversified crop pattern, while the lower area principally comprises olive groves (≈80%) and open-air vegetables (≈11%).

Table 2. Characterisation of the irrigated agriculture sector in the basin. Guadalquivir 2012.

Crops	Irrigated Area (ha)		Irrigated Area (%)		Water Use (m^3/ha)	Irrigated GM (€/ha)	Rainfed GM (€/ha)
	Upper	Lower	Upper	Lower			
Rice	38,698	0	8.98%	0.00%	10,450	787	0
Maize	16,697	2993	3.87%	0.70%	5000	1000	300
Winter cereals	64,149	11,740	14.88%	2.76%	1900	500	300
Cotton	58,813	3095	13.64%	0.73%	5000	1118	250
Sunflower	24,977	1315	5.79%	0.31%	2600	206	100
Sugar beet	12,780	673	2.96%	0.16%	4500	1765	300
Alfalfa	4950	3300	1.15%	0.78%	4500	1145	300
Vegetables (Open-Air)	35,184	46,000	8.16%	10.82%	4500	4911	250
Vegetables (Protected)	2265	0	0.53%	0.00%	4500	17,454	300
Citrus	38,476	3346	8.92%	0.79%	5400	1490	750
Grape	1650	1650	0.38%	0.39%	4000	2694	500
Olive (table)	34,644	0	8.03%	0.00%	1290	1265	400
Olive (oil)	60,920	324,510	14.13%	76.31%	1290	1480	550
Olive (intensive)	35,167	18,932	8.16%	4.45%	5000	1480	550
Almond	1800	6600	0.42%	1.55%	5000	2900	1150
Populous	0	1100	0.00%	0.26%	5400	500	400
Total	431,170	425,254	100.00%	100.00%			

Source: Authors' own based on [13].

The baseline price for irrigation is 0.06 EUR/m^3 (Table 1) with a variable tier of approximately 30% (0.02 EUR/m^3) and the rest as a flat rate. The agricultural sector's response to water pricing has been simulated by adjusting irrigated crop area (internally) and converting irrigated areas into rainfed crops when the water price causes irrigation to be halted. This is an oversimplification since certain intra-sector intra-regional water trade may be possible, but this option remains outside the scope of this analysis.

The threshold price that makes the crop unprofitable has been estimated by the algorithm shown below. The value of the threshold indicator is specific for each crop and zone. When this indicator takes a negative value, then the irrigation should be terminated. The algorithm is defined as:

$$DGM\ (Differential\ GM) = (Irrigated\ GM_{i,j} - Rainfed\ GM_{i,j}) \quad (1)$$

$$Stop\ irrigation\ when{:}\ (DGM_{i,j} - P_w Q_i) \leq 0$$

where $GM_{i,j}$ = Gross Margin of crop i in the zone j; P_w= water price; Q_i =water use per hectare of crop i. Generally, the gross margins for any agricultural crop are determined by deducting variable costs from the gross farm income of a given crop for a given period of time (usually per year or per cropping season).

2.2.2. Non-Irrigated Economic Sectors

Once the current scenario is defined, the response of the different sectors can be simulated by using known elasticities of demand for the non-irrigated economic sectors. Thanks to [15], econometric approaches to estimate price-response and allocation effects from water-pricing changes have been widely used [16,17]. Nevertheless, the estimation of the water-price elasticity faces several challenges due to the existence of artificial price systems (such as, block-rate schedules) and to the variables and dataset used, among other shortcomings [11,18].

In the specific case of the GRB, the water use (abstractions) of non-irrigated economic sectors (i.e., energy, industry, services, and livestock) represents only 5% of the total water abstractions in the GRB, while that of households amounts to 7%. In order to simplify, this method uses price-elasticity estimates as appropriate instruments to model water-use demand curves. Moreover, and in the specific case of non-irrigated sectors, water-use demand functions are estimated by incorporating the following two assumptions:

- The use of price-elasticity estimates, as given by [19] and [20]. Constant-price elasticity forms are common in water management models, and provide a proxy to estimate consumer surpluses [21];
- The calibration of isoelastic demand curves by using estimated parameters upon a single point (Price, Water use) in year 2012 (latest contrasted data available).

Price elasticities of demand can be expected to be highly inelastic for non-irrigated uses, since there are few substitutes for water use in these economic sectors [22]. Thus, in our model, water for household, industrial, and service sectors can be expected to have a marginally higher value for a certain quantity of water consumed, since each unit of water is valued much more highly than that for irrigated agriculture and much less water is consumed [7].

Table 3 summarizes the estimates for the isoelastic demand equations, as well as parameter 'K', which is obtained by solving equation (2) for current water abstraction and price for each sector.

$$Q = Kp^{\varepsilon} \tag{2}$$

Elasticities (ε) for the different sectors can be found in Table 3, and have been assumed in accordance with [19,20].

Table 3. Estimated parameters for sectoral water demand. Guadalquivir 2012.

Sectors	Elasticity (ε)	*K* (Estimated)
Livestock	−0.29	9.11
Households	−0.22	300.58
Industry (non-energy)	−0.29	70.12
Services	−0.38	80.40
Recreation	−0.29	0.34
Energy	−0.89	0.37

Source: Authors' own based on [19,20].

The elicitation of each demand curve for each sector is illustrated by the following example, which corresponds to that of the household sector. This curve is calibrated by using the pair of known values (price = 1.9 EUR/m^3, and water use = 261 Hm^3 (Table 1)) for the year 2012, and by employing the elasticity parameter (−0.22) and the estimated *K* parameter for the household sector (300.58), as shown in Table 3. In this specific case, and for the sake of simplicity, no considerations regarding disposable family income have been made. The result is an elicited demand curve for the household sector in the GRB, as defined by the following expression:

$$Q = Kp^{\varepsilon} = 300.58p^{-0.22} \tag{3}$$

Once the demand curve (water use vs. water price) is estimated for each sector, an aggregated demand curve can be obtained from the horizontal sum of all individual (or sector-specific) elicited functions. The aggregated demand curve represents the water demand for non-irrigated sectors.

2.3. Analysis of Changes in Water Use and Allocation as a Consequence of Changes in Water-Pricing Policies

Economic evaluation of simulated scenarios can provide insights into benefits and inefficiencies of alternative policy decisions at an ex-ante stage [8]. Additionally, the development of various scenarios is of value because it provides a basis for discussion and a framework for strategic planning [7]. In order to assess the global impacts of water pricing on water use and consumption in various economic sectors, price increases have been carried out by simulating the following scenarios:

- Baseline (current situation)
- Financial cost recovery (FCR)
- Financial cost recovery + environmental cost (FCR+EC)
- FCR + EC + 150%
- FRC + EC + 300%

The values for the first two scenarios can be found in [23]. Financial cost-recovery instruments can be managed by public or private agents at various stages in the provision and management of water services. In order to calculate cost-recovery rates, it necessary to estimate what income public and private agents receive for the water services they provide. Based on the standard UN System of Environmental-Economic Accounts tables, cost-recovery ratios are computed by dividing the income generated from water services (as taxes, prices, or any other financial instrument) by the cost of their provision. The financial cost-recovery (FCR) index in the GRB in 2012 based on the UN System of Environmental-Economic Accounts is estimated at 75% for agricultural and livestock economic sectors, 87% for households and services, and 91% for industry. The environmental cost (EC) is defined as the cost of damage that the various water uses impose on the environment and ecosystems. The estimation of the environmental cost (EC) is defined by the Ministry of Environment and by the values for GRB found in the aforementioned hydrological plan [13]. The EC is estimated in the GRB in 2012 with an increase of 15% above the FCR. The latter two scenarios mean major price increases (of 150% and 300% respectively above FCR + (Ministry estimated) EC) in order to analyse the impact of extreme measures of water pricing.

The impact of changes in water use by irrigation that accounts for 88% of water use is not only concentrated in agriculture but also has a multiplier effect on the rest of the economy (mainly agri-food processing, but also other complementary industries) and on services (mainly transport and service providers to farms and food processing industries), which has been simulated by using the value found for California agriculture (similar to that of Guadalquivir) of 1.49, according to [24]. Due to this multiplier effect, when agricultural GDP (irrigation) increases by 1 EUR, then the GDP of the economy as a whole grows by 1.49 EUR (i.e., an additional 0.49 for the non-agricultural sectors).

3. Results

The proposed economic model has enabled demand curves to be elicited of water abstraction vs. water price increase in the alternative scenarios analysed in this study. Figure 2 shows the integration of demand curves (water use vs. water price) of irrigated agriculture (upper and lower areas) as well as the global (integrated) demand curve of the total irrigated agriculture in the GRB. The elicited curves illustrated that the structure of the 'lower agricultural irrigated' function, integrated basically by olives and open air vegetables, passes from inelastic to elastic sections, meanwhile the function corresponding to the 'upper agricultural irrigated', with a more diversified crop pattern, shows a continuous decreasing function with little changes in the elasticity-structure of the curve.

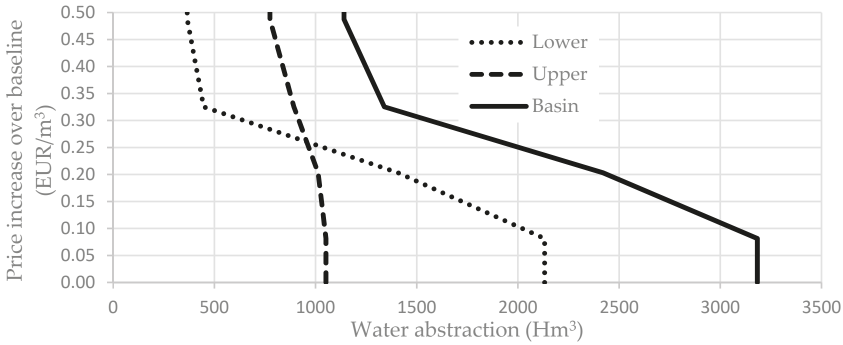

Figure 2. Elicited demand curves of water abstraction vs. water price increase (irrigation sector).

Figure 3 shows the integration of demand curves (water use vs. water price) of irrigated agriculture and the remaining economic sectors (non-irrigation), as well as the global (integrated) demand curve of the GRB. In this case, water abstraction excludes the inflow uses of energy (hydropower generation) and navigation uses. Hydropower has a lower priority in the GRB, since water is turbinated only when it is released for the interest of the other sectors, including environmental uses. Therefore, water available for hydropower is a by-product of decisions taken by the regulator in order to supply water to other sectors. In the case of navigation, this use is limited to the lower part of the GRB from the Atlantic Ocean near to Doñana National Park up to the inner-port of the city of Seville [13].

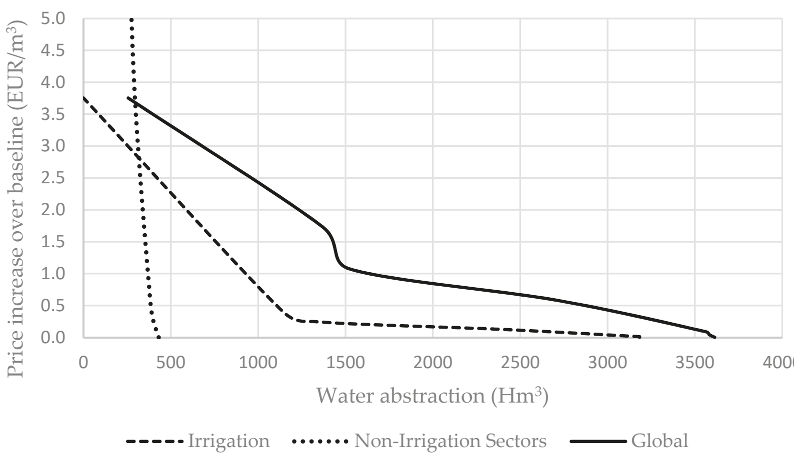

Figure 3. Elicited demand curves of water abstraction vs. water price increase (all sectors).

Based on the elicited curves, it can be clearly observed that the structure of the 'irrigated agricultural' curve passes from inelastic to elastic sections, while the curve corresponding to the remaining economic sectors (non-irrigation) shows a continuous decreasing function with minor changes in the elasticity structure of the curve.

Table 4 illustrates the response of water demand in all sectors as the water price increases as a response to the cost-recovery implementation.

Table 4. Estimated water withdrawal vs. scenarios of water pricing. Guadalquivir 2012.

	Gross Water Abstraction (hm^3)				GDP (10^6 EUR)			
	Irrigation	Non-Irrigation	Total	% Water	Agriculture	Non-Agriculture	Total GDP	% GDP
Baseline	3183	431	3614	100%	3992	60,742	64,781	100%
FCR	3183	399	3582	99%	3992	60,742	64,781	100%
FCR+EC *	3183	383	3566	99%	3992	60,789	64,828	100%
FCR+ EC * + 150%	2420	293	2713	75%	3988	60,656	64,715	100%
FCR + EC * + 300%	1266	256	1522	42%	3665	60,488	64,225	99%

Source: Authors' own. FCR = Financial Cost Recovery. EC * = Environmental cost defined by the Ministry of Environment [13].

Observation of Table 4 shows that the impact of extreme measures of water pricing reduces water abstraction by 42% vs. the baseline with the economic impact in regional GDP of a 1% reduction since agriculture (including livestock and rainfed agriculture), despite representing the sector most affected by the water pricing scenarios, constitutes only 7% of GDP. Results show that water pricing can induce water savings mainly by reducing water use in the irrigation sector although it should be considered that most of the socio-economic impact affects rural areas.

Table 5 shows the irrigated area per crop in the upper and lower areas in the various scenarios of water pricing. There is no change in the irrigation areas between the Baseline (Table 2), FCR, and FCR+EC scenarios because the increase of water pricing is insufficient to render the irrigated crops as unprofitable (inelasticity of the demand). The scenario for FCR + EC + 150% implies the substitution of crops, such as those of rice, winter cereals, sunflower, and populous, while the scenario for FCR + EC + 300% also affects maize, cotton, alfalfa, citrus, and olive (intensive) crops.

Table 5. Irrigated area per crop in the scenarios of water pricing.

Crops	Irrigated Area (ha) FCR		Irrigated Area (ha) FCR + EC *		Irrigated Area (ha) FCR + EC * + 150%		Irrigated Area (ha) FCR + EC * + 300%	
	Upper	Lower	Upper	Lower	Upper	Lower	Upper	Lower
Rice	38,698	0	38,698	0	0	0	0	0
Maize	16,697	2993	16,697	2993	16,697	2993	0	0
Winter cereals	64,149	11,740	64,149	11,740	0	0	0	0
Cotton	58,813	3095	58,813	3095	58,813	3095	0	0
Sunflower	24,977	1315	24,977	1315	0	0	0	0
Sugar beet	12,780	673	12,780	673	12,780	673	12,780	673
Alfalfa	4950	3300	4950	3300	4950	3300	0	0
Vegetables (Open-Air)	35,184	46,000	35,184	46,000	35,184	46,000	35,184	46,000
Vegetables (Protected)	2265	0	2265	0	2265	0	2265	0
Citrus	38,476	3346	38,476	3346	38,476	3346	0	0
Grape	1650	1650	1650	1650	1650	1650	1650	1650
Olive (table)	34,644	0	34,644	0	34,644	0	34,644	0
Olive (oil)	60,920	324,510	60,920	324,510	60,920	324,510	60,920	324,510
Olive (intensive)	35,167	18,932	35,167	18,932	35,167	18,932	0	0
Almond	1800	6600	1800	6600	1800	6600	1800	6600
Populous	0	1100	0	1100	0	0	0	0
Total	431,170	425,254	431,170	425,254	303,346	411,100	149,244	379,433

Source: Authors' own. FCR = Financial Cost Recovery. EC * = Environmental cost defined by the Ministry of Environment [13].

4. Discussion

A recent report by the EEA [25] acknowledges the inelastic nature of water demand in many sectors: "price does not appear to be a significant determinant of water demand". The results obtained by our study are in line with this assumption. The 'lower agricultural irrigated' function, largely comprising olives and open-air vegetables, presents elastic sections, while the function corresponding to the 'upper agricultural irrigated' scenario with a more diversified crop pattern, shows a continuously decreasing function with minor changes in the elasticity structure of the curve. The same holds true with the remaining economic sectors (non-irrigation), including the household sector. Regarding the use of water price as an instrument to induce water saving in the household sector, the EEA in its review of eight EU countries [25] concludes that: "(..) in France, Germany and Spain, the results for the household sector suggest that the prices set have a relatively minor effect on the quantity of water demanded (i.e., water demand is inelastic to price)."

The Blueprint for the water strategy document [26] follows the dominant narrative (supported by environmental NGOs, political bodies, and research institutes) in the lines: "irrigation demand is inefficient because water cost is heavily subsidized and consequently, water is too cheap. When water price increases, the demand will be reduced and then sustainability is achieved." An example of this narrative can be found in reports issued by the European Environmental Agency (2013), which include statements such as: "(. . .) increasing irrigation water prices to meet full cost recovery would maximise water use efficiency" [27] (p. 34). However, this statement contradicts the empirical observation contained in the same document, which holds that water-conserving investments depend on "incentives generated by quantity constraints and the limited role of prices" [27] (p. 43). In our study, there is no change in the irrigation abstraction between the baseline, FCR, and FCR + (Ministry estimated) EC scenarios because the increase of water pricing is insufficient to render the sector unprofitable. Major price increase scenarios (150% and 300% respectively above FCR + (Ministry estimated) EC are necessary in order to decrease the gross water abstraction for irrigation. Our results are in line with those of [28] and [29], where the authors conclude that, in the case of irrigated agriculture for moderate price increases (i.e., water cost increases to reach financial cost recovery), the response is limited, and a disproportionate price increase is necessary.

Finally, it is worth mentioning that the proposed methodology presents several limitations. One such limitation originates from the fact that no transaction costs are considered, nor are social benefits and costs that have been derived from the re-allocation of the resource, since their estimation would involve considerable difficulties [21,30], and they therefore remain outside the scope of this study. Economic models enable the economic impacts to be analysed of different management policies or decisions (e.g., water-pricing). Although it is widely accepted that no single method can capture all the dimensions associated with allocating water across all its many uses and locations at a catchment level [30], findings should be treated cautiously since there may be an inevitable gap between modelling research and its application in decision-making. This gap could be minimised by the inclusion of this type of analysis in policy assessments of a more integrated and/or holistic nature [17,31], thereby analysing policies from broader perspectives and various angles [32]. Only in this way will decision-makers attain sufficient relevant information to successfully handle decision processes.

5. Conclusions

This research focuses both on the potential impacts of water-pricing policies on water use in various economic sectors in a Southern European river basin, and on the effect that these policies incur on the re-allocation of water between alternative uses within the river basin.

The WFD [5] adopts an integrated approach to water management and grants a critical role to economic instruments, such as the use of "water pricing" and "full cost recovery" (Article 9), as efficient measures to achieve environmental objectives. However, this study concludes that the role of prices remains limited regarding water-use reduction although it does remain a key instrument for achieving cost recovery for water services to ensure the maintenance and financing of existing and future water infrastructure.

The exploratory model developed herein may serve policy makers in their assessment of the potential effects of water-pricing policies on the water used and on consumption from an inter-sector approach.

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water

MDPI

Article

Analysis of the Dynamic Changes of the Baiyangdian Lake Surface Based on a Complex Water Extraction Method

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Abstract: Lakes have an important role in human life and the ecological environment, but they are easily affected by human activity and climate change, especially around urban areas. Hence, it is critical to extract water with a high precision method and monitor long-term sequence dynamic changes in lakes. As the greatest natural lake of the Beijing-Tianjin-Hebei region, Baiyangdian Lake has a significant function in human life, socio-economic development, and regional ecological balance. This lake area has shown large changes due to human activity and climate change. The change monitoring process of the water surface is of great significance in providing support for the management and protection of the lake. The Spectrum Matching based on Discrete Particle Swarm Optimization (SMDPSO) method is a new, robust, and low-cost method for water extraction, that has obvious advantages in extracting complex water surfaces. In this paper, the SMDPSO method was used to extract the water surface of Baiyangdian Lake by Landsat images from 1984 to 2018. This method has a good effect on complex water surface extraction with vegetation, shadows, and so forth, and the Landsat images have higher resolution and longer time series. The main contents and results of this paper are as follows: (1) We verified the applicability of the SMDPSO method in the Baiyangdian Lake using visual interpretation and correlation analysis. The relative errors between observed and extracted results were all less than 5% in spring, summer, and fall, and the correlation coefficient between the water area and water level was 0.96. (2) According to seasonal verification and comparison of the extraction results, the SMDPSO method was used to extract the water surface area of Baiyangdian Lake during spring of the years 1984–2018. Water area changes of Baiyangdian Lake can be divided into four periods: Dry period (1984–1988), degraded period (1989–2000), stable period (2000–2008), and recovery period (2008–2018). The water area reached a maximum of 280 km^2 in 1989 and a minimum of 44 km^2 in 2002. (3) The possible causes of the changes in the water area of Baiyangdian Lake were also analyzed. The changes were caused by climate and human activities during the first and second periods, but mainly human activities during the third and fourth periods. In fact, effective policies combined with water conservancy projects were directly conducive to improving or even recovering the water and ecological environment of Baiyangdian Lake. Considering its importance for the benign development of the Beijing-Tianjin-Hebei Region and the construction of the Xiong'an New Area, a policy is necessary to ensure that the lake's ecological environment will not be destroyed under the premise of economic development.

Keywords: Baiyangdian Lake; Landsat; complex water extraction; SMDPSO; dynamic changes

1. Introduction

Lakes are important parts of the hydrosphere, which not only supply water resources needed by humans and the ecological environment, but also maintain the climate system and water cycle [1–4]. Lakes are very sensitive to climate change or human activities [5]. In recent decades, many lakes have shrunk significantly due to intensive human activities and climate change [6,7]. Moreover, shrinking and drying of lakes has aggravated the deterioration of the regional environment and directly threatened the livelihoods of local people [8].

Data from remote sensing satellites, such as Landsat, MODIS, Sentinel-1, CBERS-1, and so on [9–12], has been widely used for mapping surface water. Landsat datasets are likely the most common data employed to identify the water surface because of their high spatial resolution, free availability, and their long sequence feature [10,13]. Han et al. [11] used Landsat data to study changes in the winter wetlands of Poyang Lake from 1973 to 2013. Almost simultaneously, Donchyts et al. [12] and Pekel et al. [14] both detected changes in global surface water over three decades using most of the Landsat images that had been produced before 2017. Yang et al. [15] used Landsat and Huanjing (HJ) satellite data to monitor the dynamics of lakes in the Changtang Plateau. Landsat data is very suitable for studying long-term changes in terrestrial water, especially for lakes.

An efficient water extraction method is critical to obtaining accurate water surface area. Three main methods can be summarized for water body extraction based on the current, common methods: (1) Threshold methods based on a related water body index; (2) supervised and unsupervised classification, including machine learning algorithms; and (3) water-specific classification methods, such as HSV (Hue, Saturation, and Value) [14] and Spectrum Matching based on Discrete Particle Swarm Optimization (SMDPSO) [16]. The water index based on the spectral curve of the water body is the simplest method for water body extraction, with examples being Normalized Difference Water Index (NDWI) and Modified Normalized Difference Water Index (MNDWI). The appropriate band is selected to construct the model, and then the appropriate threshold is selected to extract the water body [16,17]. There are many types of water indices, and NDWI and MNDWI are the two most commonly used indices. These water indices are easy to calculate and less time consuming, so researchers often use them to extract water information. However, difficulties in threshold determination and the great instability in the classification results are the main limitations of this method. Machine learning for supervised classification is an effective way to save time and labor as well as maintaining high accuracy, with examples being support vector machine (SVM) and random forest (RF) [18]. Mueller et al. [19] used a regression tree model for sample training to detect water information in Australia. Rao et al. [20] used the RF classifier for sample training to extract surface water in the Yangtze River Basin. The accuracy of machine learning for supervised classification seems to be better than the water body index. However, machine learning algorithms are subject to the selection of sample points, and have high requirements for sample points and bands during the classification process, which would face great difficulty in long-term sequence water extraction, and may cause large errors in classification results. Water-specific classification methods are experience-based optimization algorithms with simple operations, good effects, and strong applicability. These methods are based on prior knowledge, require fewer parameters, and are more highly automated than supervised or unsupervised classification. Pekel et al. [21] proposed a near real-time water surface detection method based on HSV (Hue, Saturation, and Value) transformation of MODIS multi-spectral time series data, and also used this method for mapping global water surface bodies by Landsat datasets [14]. Jia et al. [16] proposed a SMDPSO method for complex water extraction, and tested the efficiency of the method through the eight typical global regions. The results showed that the water extraction accuracy and robustness of the SMDPSO method are better than those of the water body index and supervised classification methods.

Currently, correlation analysis is one of the ways to explore possible causes. Tao et al. [8] explored the causes of lake water losses by analyzing the correlation between lake area on the Mongolian Plateau and rainfall, coal production, and agricultural irrigation area. Zhang et al. [22] analyzed the relationship between rainfall and wetland areas and the area of pure water in Baiyangdian Lake to explore the impact of rainfall on wetland area changes. Therefore, the possible causes for changes in the Baiyangdian Lake water area are qualitatively analyzed based on correlation analysis.

Baiyangdian Lake is an important part of the Xiong'an New Area, which is being developed into a national-level new area, like the Shenzhen and Pudong New Areas, to solve the "big city problem" in Beijing, and promote the coordinated development of the Beijing-Tianjin-Hebei Urban Agglomeration. Approximately one-fifth of the area in the Xiong'an New Area is a part of Baiyangdian Lake. Baiyangdian Lake has a key role in creating a green and ecologically livable Xiong'an New Area. Since the 1970s, under the combined effect of climate change and human activity, Baiyangdian Lake has experienced many periods with little water [23–26]. According to the hydrological record between 1919 and 1965, there was almost no water in the spring of 1922. More importantly, in the years from 1965 to 2005, the frequency of no water in Baiyangdian Lake increased [27]. The emergence of these problems has a great impact on Baiyangdian Lake and its surrounding environment. Therefore, it is significant to recognize the past changes in the water surface area of Baiyangdian Lake and the current situation to protect the ecological environment of Baiyangdian Lake and promote the efficient construction of the Xiong'an New Area.

Based on the above, the purpose of this paper is: (1) To accurately extract the water surface area of Baiyangdian Lake and verify the accuracy of SMDPSO, (2) to acknowledge the historical changes and current status of the water surface area of Baiyangdian Lake, and (3) to analyze the causes of changes in the lake water surface area based on climate change and human activities.

2. Materials and Methods

2.1. Study Area

Baiyangdian Lake, with a total area of 366 km^2 [28] (38°43′–39°02′ N and 115°38′–116°07′ E), is located in the central area of the Hebei Province, China (Figure 1). It mainly belongs to Anxin County and is surrounded by the counties of Rongcheng, Xiongxian, and Gaoyang, and the city of Renqiu [29]. The warm semi-arid climate characteristic is typical in this area, where the annual average temperature is 7.3–12.7 °C, and the annual average precipitation, mainly concentrated between June and September, is about 563.9 mm [22]. However, the annual average evaporation is approximately 1369 mm, which is much greater than the amount of precipitation [30]. If there is no manual intervention, the lake will become dry more and more. Therefore, a series of water conservancy projects have provided water to the Lake, including the Yellow River Diversion Project and the Central Line Project of the South-to-North Water Diversion.

Baiyangdian Lake is the largest freshwater lake in the Beijing-Tianjin-Hebei urban agglomeration [31]. It has an import function to maintain the ecological balance in the region, and plays a key role in the protection of biodiversity and rare species resources [24]. More than 3700 ditches and reed areas divide the whole Baiyangdian Lake into several lakes of different sizes, and numerous lakes connected by gullies create many small islands on which there are villages and gardens [32].

Considering the complex water of Baiyangdian Lake, we divided the Baiyangdian Lake into seven subregions based on the functional area division of Baiyangdian Lake [33], combined with the distribution and integrity of lakes.

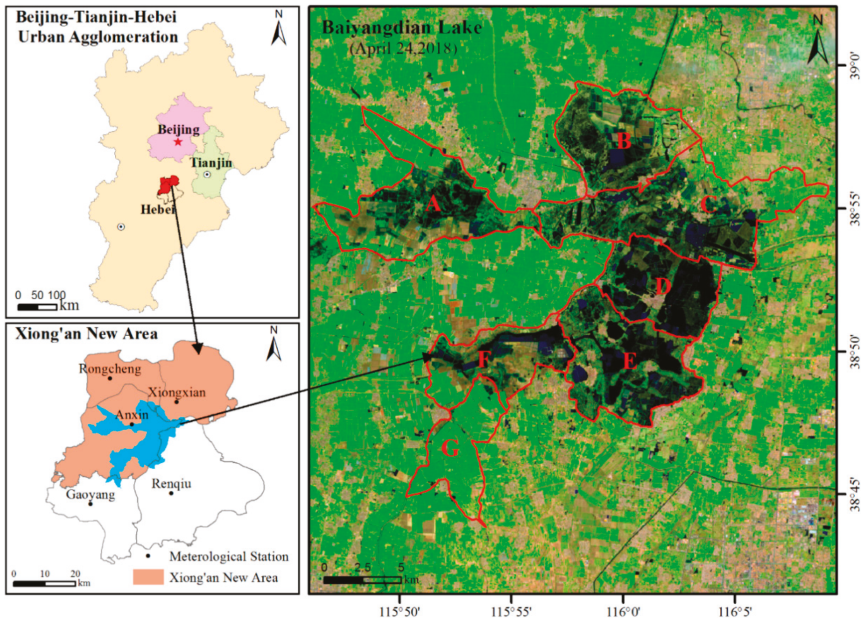

Figure 1. Location of the study area (the remote sensing image of Baiyangdian Lake is displaced by false color images and using bands 7, 5 and 3).

2.2. Materials

2.2.1. Landsat Time Series

In this study, Landsat images were used to extract the water surface area of Baiyangdian Lake. Remote sensing images with less than 20% cloud cover from 1984–2018 were downloaded, and were mainly derived from Landsat 5 Thematic Mapper (TM), 7 Enhanced Thematic Mapper (ETM), and 8 Operational Land Imager (OLI). We chose the images from spring (March, April, May), summer (June, July, August), fall (September, October, November), and winter (December, January, February) of 1990, 2001 and 2017 to determine which season presents the best water extraction effect. Comparing the four seasons, the water body extraction was less affected by vegetation, and the water surface area generally did not change greatly, in the spring (March, April and May), which is a non-rainy season in the study area. Therefore, cloudless images during the spring of the years 1984–2018 were selected in this study. Detailed information on the Landsat images used in this study is shown in Figure 2. There are 37 Landsat images used in this study, and the images were downloaded from USGS/EROS (http://landsat.usgs.gov/).

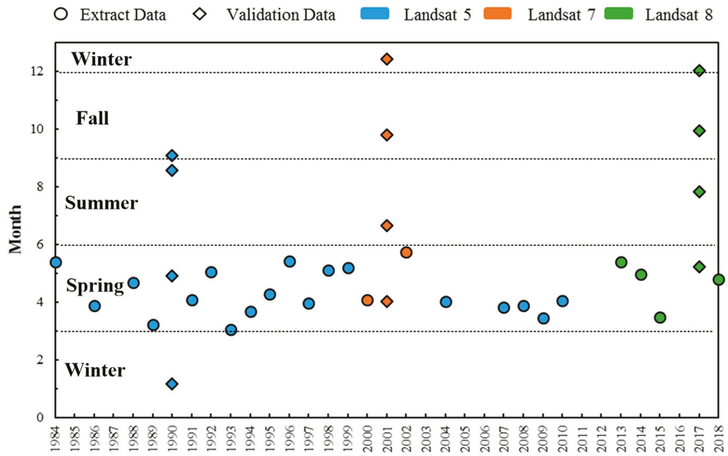

Figure 2. Temporal distribution of Landsat images used in this study.

2.2.2. Hydrological and Climate Data

In this study, water level (WL), annual precipitation (AP), annual average temperature (AAT), natural inflow (NI), and water diversion volume (WDV) data was used to analyze the possible effects on water surface area changes.

The WL data was used for correlation analysis with the extracted water area data to determine the water surface area extraction accuracy, and this data was derived from the county annals of Anxin from 1984 to 2008. The average WL per month was obtained from the data of three hydrological stations in South Liuzhuang, Duancun, and Wangjiazhai, near Baiyangdian Lake. WL data from 2010 to 2015 was derived from the hydrological data yearbook of the Daqing River Basin each year, and this data was obtained from the three hydrological stations of Anxin, Duancun, and Wangjiazhai.

NI refers to the surface runoff, underground runoff, rainfall, and so on, entering Baiyangdian Lake, and does not include artificially supplemented water. WDV is the amount of water that transfers from reservoirs, rivers, and so on, into Baiyangdian Lake by artificial means. NI and WDV were also used to analyze causes for water area changes. The data of NI (1984–2010) and WDV (1984–2010) (Figure 3) was acquired from Cui et al. [28]. The rest of the WDV data was from the Water Conservancy Briefing of the Baoding Water-Control Bureau.

Precipitation and temperature data (1984–2017) from five stations (stations of Anxin, Gaoyang, Rongcheng, Xiongxian, and Renqiu) around Baiyangdian Lake was acquired from the China Meteorological Administration (Figure 1), and was used to analyze the effects of rainfall and temperature on the water area of Baiyangdian Lake (Figure 3). In this paper, AP and AAT used the average of five stations.

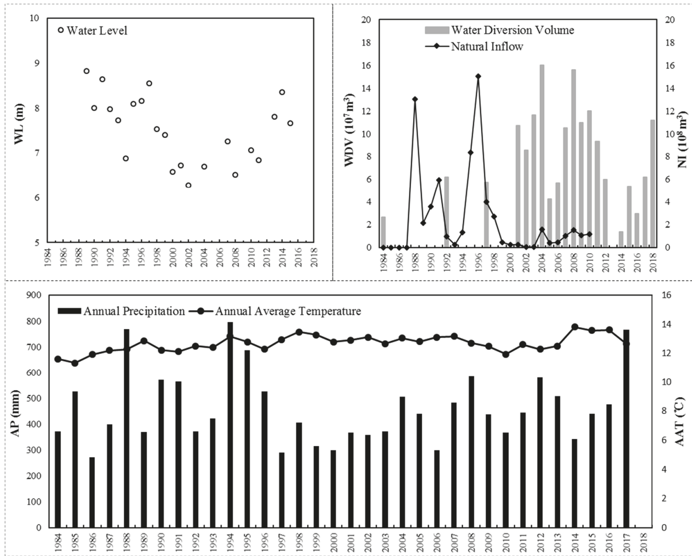

Figure 3. Hydrological and climate data used in this study.

2.3. Methods

2.3.1. SMDPSO Method

SMDPSO is a new, simple, and high-precision water extraction algorithm. It has fewer input parameters, lower cost, and higher stability and accuracy than supervised and non-supervised classification, especially presenting clear advantages in complex water extraction [16]. The algorithm consists of two main steps (Figure 4): (1) Obtaining water probability maps from multispectral remote sensing data through appropriate spectral matching methods, and (2) identifying water pixels from the water probability image by using the discrete particle swarm optimization (DPSO) analytical objective function.

Water probability can be calculated using Equation (1). In this study, we use the standard water spectrum for Landsat 8 images, which come from Jia et al. [16], to calculate water probability and apply it to Landsat 5 and 7 images.

$$P_w = \cos\left(\vec{W}, \vec{O}\right) \cdot \text{dist}\left(\vec{W}, \vec{O}\right) \tag{1}$$

$$\cos\left(\vec{W}, \vec{O}\right) = \frac{\vec{W} \cdot \vec{O}}{\|\vec{W}\| \cdot \|\vec{O}\|} \tag{2}$$

$$\text{dist}\left(\vec{W}, \vec{O}\right) = 1 - \frac{1}{\sqrt{b}} \sqrt{\sum_{i=0}^{b} (w_i - o_i)^2} \tag{3}$$

where $\vec{W} = (w_1, w_2, \cdots w_b)$ and $\vec{O} = (o_1, o_2, \cdots o_b)$ represent the spectral vector (i.e., the spectral curve) of typical water and ground objects, respectively, and b is the number of bands. The cosine similarity (Equation (2)) and distance similarity (Equation (3)) are values between 0 and 1. Thus, the water probability is in the range of [0,1]. A higher value of P_w indicates a greater probability of water.

The next step is to construct the objective function of the water classification. The objective function is as follows (Equation (4)):

$$T = c_1 \sum_{k=1}^{rows \times cols} P_{w,k} + c_2 \sum_{k=1}^{rows \times cols} P_{nw,k} - c_3 \frac{\overline{D}_{nearest}}{\sqrt{rows^2 + cols^2}} \tag{4}$$

where c_1, c_2, and c_3 are constants, are calculated according to Jia et al. [16], and represent the weight of the water portion, no water portion, and neighborhood; P_w is the water probability when the pixel is water; P_{nw} is the nonwater probability when the pixel is nonwater; and $\overline{D}_{nearest}$ is the nearest distance from one water pixel to another water pixel at the same tile (Equation (5)). Images are divided into tiles with *rows* $\times$ columns (*cols*).

$$\overline{D}_{nearest} = \begin{cases} 0, \ No \ water \\ \sqrt{rows^2 + cols^2}, \ Number \ of \ water \ is \ equal \ to \ 1 \\ Nearest \ distance \ from \ one \ pixel \ to \ another \ otherwise \end{cases} \tag{5}$$

The pixel is classified as water or nonwater by maximizing the objective function. The key of SMDPSO is using DPSO to solve the objective function and obtain the classification result. DPSO is an optimization algorithm that resolves problems in discrete space [34].

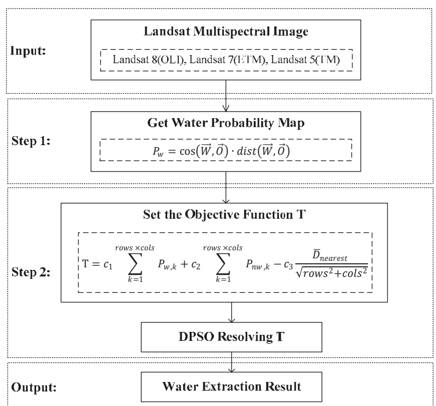

Figure 4. Steps of extracting the water surface area using the SMDPSO method.

2.3.2. Seasonal Verification of SMDPSO

In this study, we aim to verify the accuracy of water extraction in each season and select the best season for long-term sequence analysis. Considering the complex water characteristics of Baiyangdian Lake, the overall accuracy was impossible to calculate accurately, because of too many water plaques. Therefore, we selected three typical small water plaques, which were used to test the results. April, August, September, and January of 1990 were selected to represent the four seasons, and D1 (Figure 5) was used as the sample area to analyze the error between the extracted and the observed water areas. Similarly, April, June, September, and December of 2001 represent the four seasons, and C1 (Figure 5) was taken as the sample area. The images of May, July, September, and December of 2017 represent the four seasons of spring, summer, fall, and winter, and E1 (Figure 5) was selected as the sample area. We manually interpreted the water surface area of D1, C1, and E1 during the four seasons as the actual water areas, and used the following formula (Equation (6)) to calculate the error between the extracted water area and the observed water area.

$$R = (|S_e - S_a| / S_a) \times 100\% \tag{6}$$

where R is the relative error between the water area of the extracted and the observed, S_e is the area of the extracted water area, and S_a is the area of the observed water area.

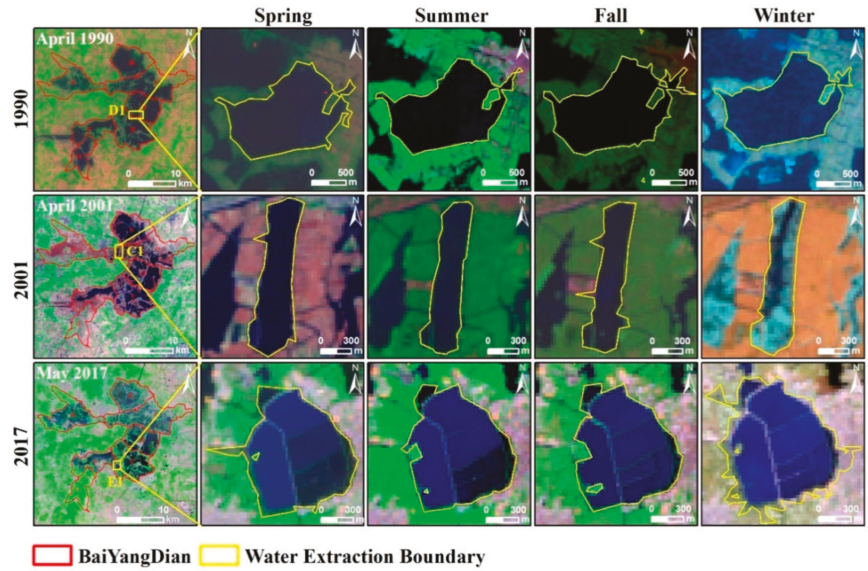

Figure 5. Seasonal verification of SMDPSO with selected data and sample areas. D1, C1, and E1 were the sample areas of 1990, 2001, and 2017, respectively. All images were displaced by false color images (bands 6, 4, and 2 for 1990 and 2001; and bands 7, 5, and 3 for 2017).

2.3.3. Pearson Correlation Coefficient and Water Inundation Frequency

Pearson correlation coefficient (r) (Equation (7)) is an effective tool for measuring the closeness of two random variables X and Y. The range of r is -1 to 1, greater than 0 indicates a positive correlation, and less than 0 indicates a negative correlation. The correlation is greater when its absolute value is close to 1 [35]. The general situation will be expressed as 5 intervals of 0~0.2, 0.2~0.4, 0.4~0.6, 0.6~0.8, and 0.8~1.0, which represent without correlation, weak correlation, medium correlation, strong correlation, and extremely strong correlation, respectively [36].

$$r = \frac{\sum_{i=1}^{n}(X_i - \overline{X})(Y_i - \overline{Y})}{\sqrt{\sum_{i=1}^{n}(X_i - \overline{X})^2 \sum_{i=1}^{n}(Y_i - \overline{Y})^2}} \tag{7}$$

where r is the correlation coefficient of variables X and Y, X_i is the water area of Baiyangdian Lake every year, and Y_i is other data (WL, AP, AAT, and NI) related to water area each year. $\overline{X}$ and $\overline{Y}$ are the means of X and Y. n is the sample size.

Water inundation frequency (WIF) can reflect the spatial characteristics of Baiyangdian Lake. In this study, WIF was calculated by the number of times a pixel is flagged as a water body divided by the number of all observations per pixel. In this paper, a total of 37 images were used, so the number of all observations per pixel is 37. According to Rao et al. [20], it was named as permanent water when the WIF is greater than 60%, otherwise it was named as temporary water.

3. Results

3.1. Verification of Water Extraction Results

3.1.1. Comparison and Verification for Different Seasonal Results

By visually comparing the extracted water surface area with the original remote sensing image, the overall effect of the SMDPSO is adequate (Figures 6–8). In the spring, summer, and fall seasons, the degree of matching of the extracted water surface area with the original image is very high, but the water surface extracted in the winter has an overlarge phenomenon (Figures 6d, 7d, 8d). The reason for the excess in winter may be due to ice, snow, and wet soil near the water. SMDPSO is also applicable to Landsat 5 and 7 through verification.

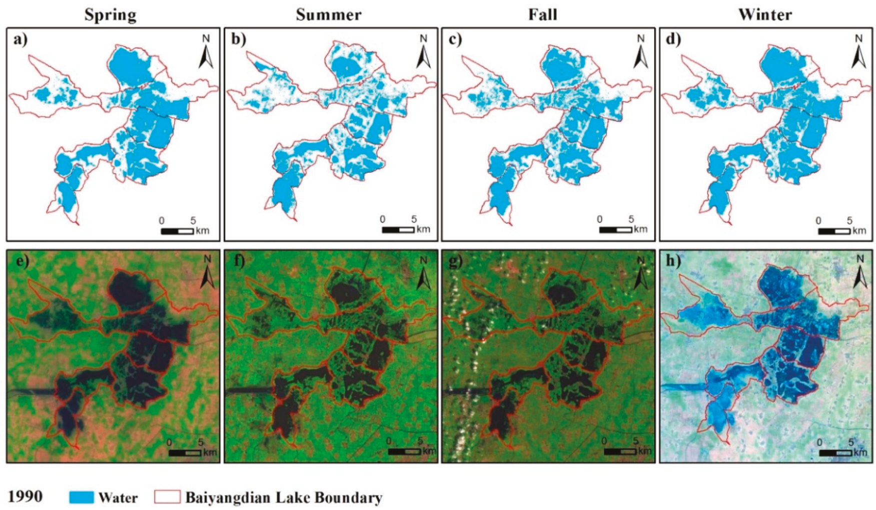

Figure 6. Comparison of water extraction results and original images for all four seasons in 1990. Extracted water surface in the (**a**) spring, (**b**) summer, (**c**) fall, and (**d**) winter of 1990. False color images (bands: 6, 4, and 2) of (**e**) spring, (**f**) summer, (**g**) fall, and (**h**) winter in 1990.

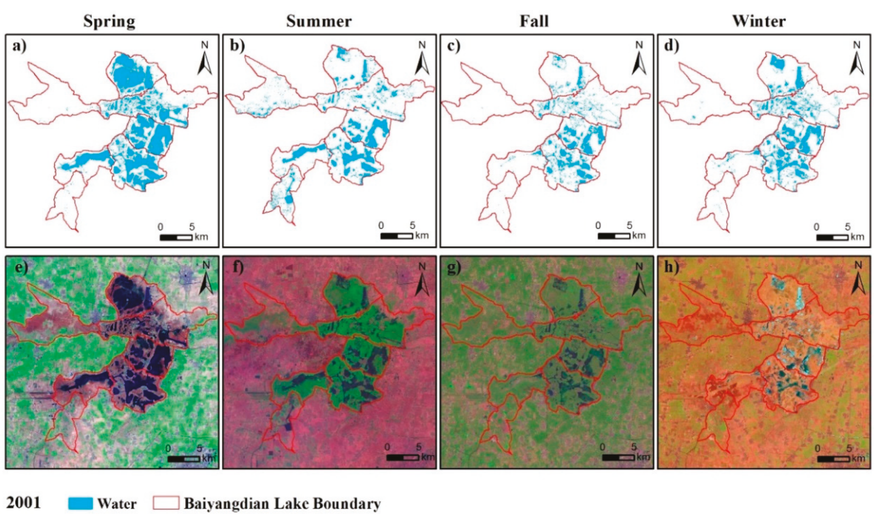

Figure 7. Comparison of water extraction results and original images for all four seasons in 2001. Extracted water surface in the (**a**) spring, (**b**) summer, (**c**) fall, and (**d**) winter of 2001. False color images (bands: 6, 4, and 2) of (**e**) spring, (**f**) summer, (**g**) fall, and (**h**) winter in 2001.

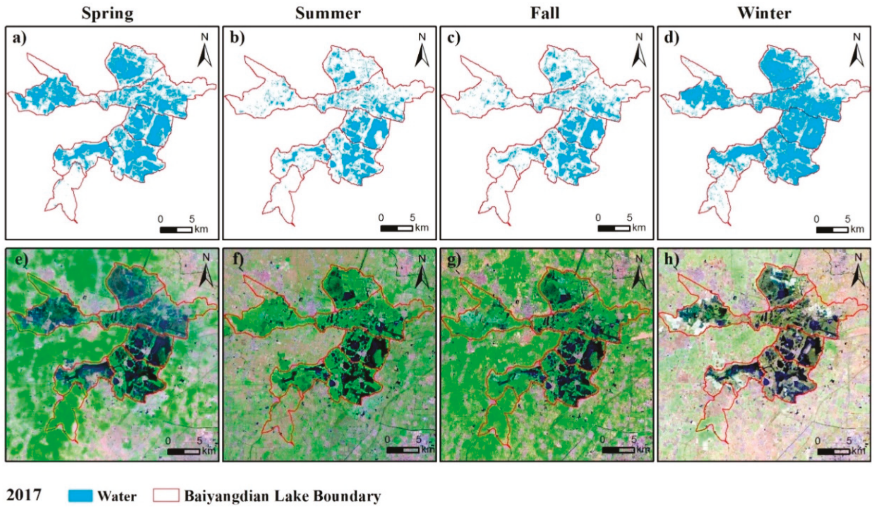

Figure 8. Comparison of water extraction results and original images for all four seasons in 2017. Extracted water surface in the (**a**) spring, (**b**) summer, (**c**) fall, and (**d**) winter of 2017. False color images (bands: 7, 5 and 3) of (**e**) spring, (**f**) summer, (**g**) fall, and (**h**) winter in 2017.

We calculated the *R* between the water area of the extracted and the observed in the three sample areas of D1, C1, and E1 (Figure 5). Based on the details of Figure 5, the boundary lines of the water extraction results in the three sample areas agree well with the original images, but several water extraction results look larger than the actual values in the winter. The *R* values of these results are conducted in Figure 9, showing that the water extraction accuracy of most images is satisfying where *R* is less than 5%, with only the images in the winter of 2017 having a relatively large error. The mean and maximum value of *R* is 0.9% and 1.7% in spring, 3% and 3.8% in summer, 2.2% and 3.1% in fall, and 8.3% and 20.4% in winter, respectively, indicating the water extraction results are more accurate and reliable in spring than other seasons. Therefore, in this paper, the water extraction results from the remote sensing images in the spring are used to analyze the changes in Baiyangdian Lake from 1984 to 2018.

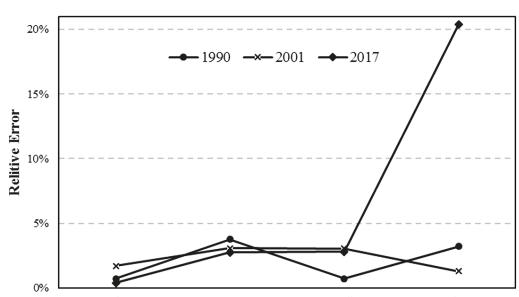

Figure 9. The relative error between the extracted water area and the actual water area.

3.1.2. Comparison with Water Level Data

In this study, WL data was used to test the water extraction accuracy of Baiyangdian Lake, and the month of the WL data corresponds to the month of the remote sensing images. The data was used to calculate the *r* between WL and water area.

Baiyangdian Lake is a low-lying land located in the alluvial plain, and the change of water surface area will obviously cause the water level to change. Figure 10 shows that there is a significant positive correlation between the WL and water area (r = 0.96, $p < 0.01$). It also suggests that the results of the water surface extraction are very accurate, and the SMDPSO method has good applicability to Baiyangdian Lake.

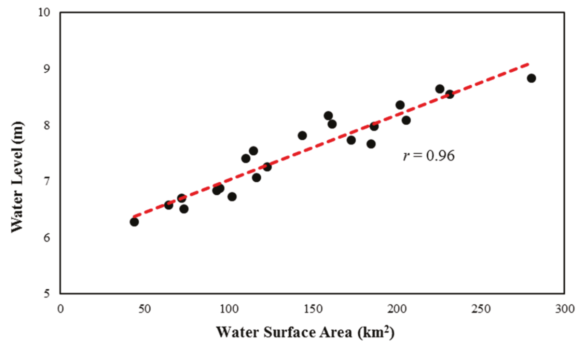

Figure 10. The relationship between the Baiyangdian Lake water area and water level.

3.2. Variation Characteristics of the Water Area in Baiyangdian Lake

3.2.1. Interannual Change Analysis of Baiyangdian Lake

From 1984 to 2018, the water area in Baiyangdian Lake was divided into four periods (Figure 11). First, Baiyangdian Lake was very water-deficient from 1984–1988 and the average water area was 11 km^2. Due to extremely heavy rain, the dry state of Baiyangdian Lake ended, and the water area reached its largest area in 1989, with an area of 280 km^2. The second phase was a degradation period, and the water area of Baiyangdian Lake gradually decreased from 1989 to 2000. In 2000, the water area dropped to 64.15 km^2, with an average annual reduction of approximately 11 km^2. The third phase (2000–2008) was a relatively stable period, and there was no significant increase or decrease in the water area. Finally, the fourth phase was a restoration period (2008–2018), and the water area of the lake increased approximately 12 km^2 per year during this period.

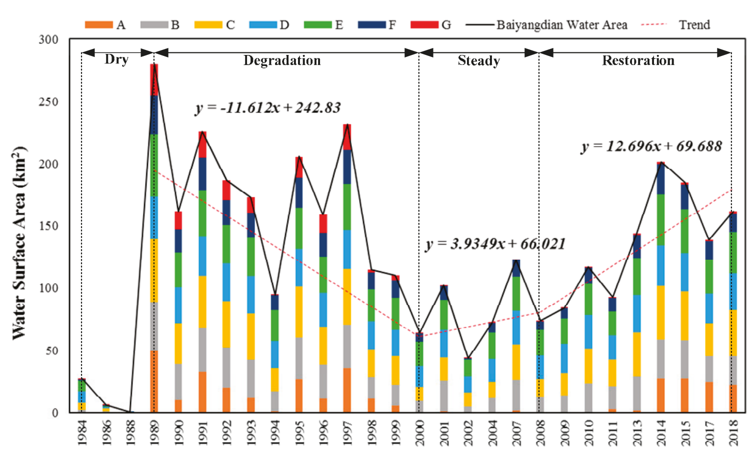

Figure 11. Water area changes in Baiyangdian Lake and 7 subregions from 1984 to 2018.

The area changes in several subregions were like the whole lake, except for A and G. The water area of G reached the largest value with a total of 25 km^2 in 1989, but the water in G was almost non-existent after 1999. Similarly, the water area of A also reached a high level (approximately 49.93 km^2) in 1989. After 2000, the surface water almost disappeared. However, in 2014 the water area was restored to 27.37 km^2. The water area of A after the recovery period was only approximately half of the result during the maximum period.

Four periods can be clearly divided into water surface area variations in Baiyangdian Lake, as shown in Figure 12. The water surface area in 1989, 1995, 2014, and 2018 was clearly large and covers a wide area. The water in A had gradually degraded since 1997 and began to recover in 2014, but the range of this water has been significantly reduced compared to the largest water area period in 1989. There is still no water in G and the western part of F, because this water area was completely converted into farmland, although most the water of each subregion recovered in 2018. Due to the water surface area changes in Baiyangdian Lake from 1984 to 2018, the degree of water fragmentation has become increasingly serious. The water surface area was complete and smooth in 1989, while fragmented and messy in 2018.

Figure 12. Spatial variations in water area from 1984 to 2018 in Baiyangdian Lake.

3.2.2. Spatial Change Analysis of Baiyangdian Lake

To study the spatial changes of lake surface, we calculated the WIF and divided it into five grades: 0–20%, 20–40%, 40–60%, 60–80%, and 80–100% (Figure 13). The permanent water mainly concentrated in the five subregions of B, C, D, E, and F. In B and F, a complete small lake shape can be seen, and D and E are composed of several small lakes, but C is different from B, D, E, and F, as it has no complete lake shape. The temporary water is concentrated in A and G, especially the parts where the WIF is less than 40%. It can be regarded as no water if the WIF is less than 20%, because the distribution of non-water is consistent with farmland, villages, and so on.

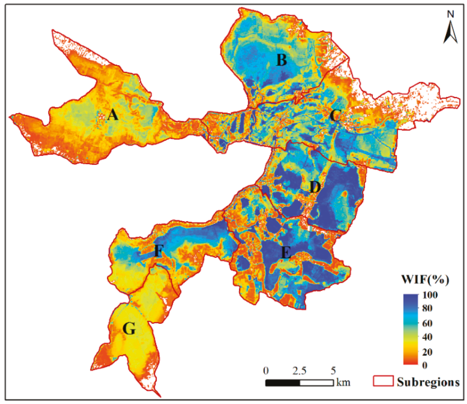

Figure 13. Water Inundation Frequency (WIF) map of Baiyangdian Lake.

We calculated the water area percentage for each subregion out of the total water area beginning in 1989, because the lake had almost no water in 1984–1988 (Figure 14). The water areas of C, D and E occupied approximately 60% of the total water area, and reached approximately 80% during 2000–2008 when the water area was small. The differences in water area between different subregions of Baiyangdian Lake were small in 1989, 1990, 1995, 1997, and 2014, which were the periods that the water area was large. In the year when the water area was small, the areas with water were mainly concentrated in B, C, D, E and F, while the other subregions had little or no water. This also shows that the water in Baiyangdian Lake was mainly located in the five subregions of B, C, D, E and F.

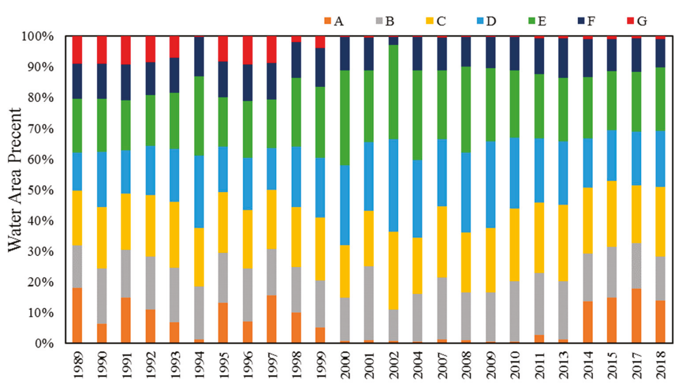

Figure 14. Each subregion as a percentage of the total area. For reference, the percentage of A refers to the water area of a divided by the water area of the entire Baiyangdian Lake in 1989.

3.3. Possible Causes for Changes in Water Area in Baiyangdian Lake

3.3.1. Temporal Changes

From 1984 to 1988, Baiyangdian Lake was in a dry state. This dry state of Baiyangdian Lake was caused by the combined effects of climate and human activities [37]. The area of cultivated land increased continuously after the land reform in 1980, so agricultural water consumption increased significantly [27]. Moreover, a large number of reservoirs and dams were built upstream of Baiyangdian Lake [38]. Simultaneously, the AP was less, and the NI was zero during this period. There were large consumption and no water supply, so Baiyangdian Lake was in a dry state.

The water area of Baiyangdian Lake showed a decreasing trend from 1989 to 2000. We separately calculated the Pearson correlation coefficients between the water area and AP, AAT, and NI during this period. There is no positive correlation between AP and the water area, and it is positively correlated with NI, and negatively correlated with AAT (Figure 15). The reduction of NI and the increase of AAT have certain impacts on the reduction of water surface area, but climate change was not the only cause. The impact of human activities can be inferred to be the most important cause for the reduction in the water area of Baiyangdian Lake. The possible reason was agricultural irrigation. Baiyangdian Lake is part of the plain, and the surrounding agriculture is relatively developed. In studying the driving force of Baiyangdian landscape change, Zhuang et al. [39] pointed out that the grain output of the Baiyangdian basin increased from 2.6 $\times$ 10^9 kg in 1970 to 6.6 $\times$ 10^9 kg in 2007, and the agricultural water consumption in this basin accounted for 78.8% of the total water consumption in 2006.

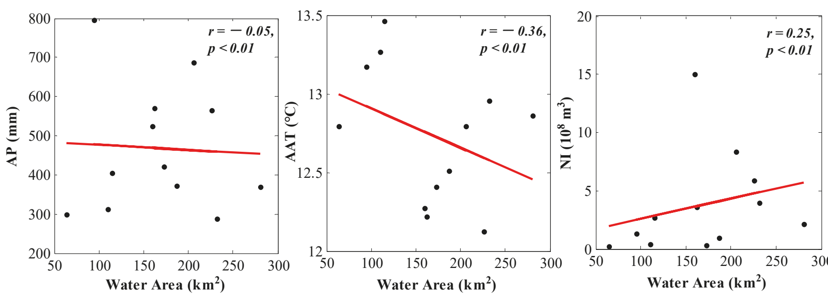

Figure 15. The Pearson correlation coefficient between water area and annual average temperature (AAT), annual precipitation (AP), and natural inflow (NI) from 1989–2000.

The water area of Baiyangdian Lake did not appear to obviously increase or decrease from 2000 to 2008. We calculated the Pearson correlation coefficients between the water area and AP and AAT. The calculation results show that precipitation has a certain influence on the stability of the water surface area, but temperature does not (Figure 16). To ensure that the water level of Baiyangdian Lake is not lower than the warning value, the government maintains the stability of the water surface area by water diversion. From 2000 to 2008, approximately 0.83 billion m^3 of water entered Baiyangdian Lake. After completion of the Conducting Yellow River Water to Baiyangdian Lake Project in 2006, the water diversion to Baiyangdian Lake was mainly from the Yellow River, while previously water came from the surrounding reservoirs.

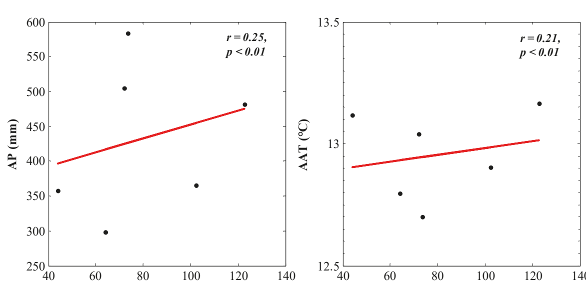

Figure 16. The Pearson correlation coefficient between water area and AAT and AP from 2000–2008.

From 2008 to 2018, the water area of Baiyangdian Lake showed an upward trend. We also calculated the Pearson correlation coefficient between the water area and AP and AAT during this period. We were surprised to find that the change in water area is negatively correlated with precipitation and positively correlated with temperature (Figure 17). The increase in the water area has little to do with precipitation and temperature. Therefore, the development of artificial water diversion projects may be the main cause for the increase in water area. The Conducting Yellow River Water to Baiyangdian Lake Project is the main means of supplementing water to Baiyangdian Lake. In addition, the Central Line Project of South-to-North Water Diversion officially passed supplemental water in 2014, and brought much of the ecological water to Baiyangdian Lake. In recent years, a series of policies on ecological protection have been introduced that played a leading role in the restoration of the ecological environment of Baiyangdian Lake. In particular, the government stressed that we must vigorously promote the construction of ecological civilization and reverse the deterioration of the ecological environment in 2012. There was 0.83 billion m^3 of water that entered Baiyangdian Lake from 2000–2008, and 0.66 billion m^3 of water entered after 2008. However, the mean value of water area from 2000–2008 was 80 km^2, and 141 km^2 from 2009–2018. This shows that effective policies can greatly contribute to the restoration of the water surface area of Baiyangdian Lake.

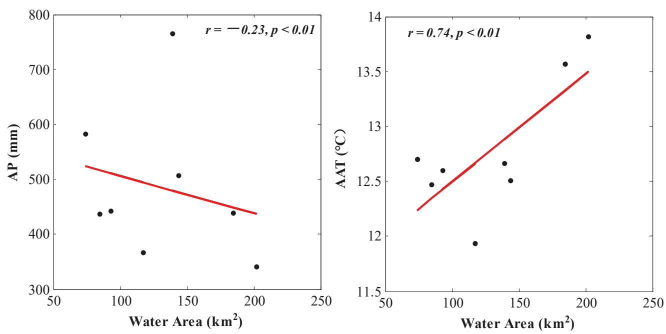

Figure 17. The Pearson correlation coefficient between water area and AAT and AP from 2008–2018.

3.3.2. Implications of Policies on the Water Surface Area

The possible causes for water surface area changes were quite different in four periods (Figure 18). From 1984 to 1988, the reduction of precipitation and natural inflow, and the increase in agricultural irrigation, led to the absence of water in Baiyangdian Lake. Heavy rain in 1988 caused the largest natural inflow and restored the lake to its largest water area during years 1984–1988 (Figure 3). From 1989 to 2000, precipitation, natural inflow, annual average temperature, and agricultural irrigation were possible reasons for the decline in water area during this period. In addition, people's environmental awareness was relatively weak during this time, and the government did not carry out large-scale projects for water diversion to rescue Baiyangdian Lake. From 2000 to 2008, precipitation, reservoir water diversion, and Yellow River diversion were possible causes ensuring the water surface area remained unchanged, and water diversion was the main cause. However, this maintenance consumes large amounts of manpower and material resources, and is not a long-term solution. From 2008 to 2018, Yellow River diversion, South-to-North Water diversion, and effective policies were the main reasons for the increased water surface area. The future water surface area of Baiyangdian Lake will increase under policy protection, and Gu et al. [26] proposed paying attention to flood prevention in the Baiyangdian area.

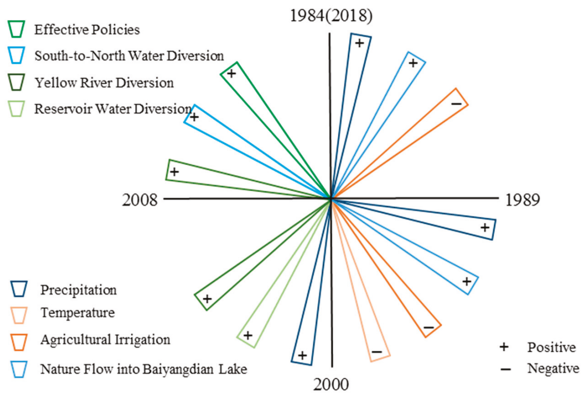

Figure 18. The possible causes for the change in water area in Baiyangdian Lake during the four periods. + indicates that this factor is positively correlated with the change in water area, and − indicates that the factor is negatively correlated with the change in water area.

4. Discussion

4.1. Comparison with Other Studies in Baiyangdian Lake

Comparing the variation in the water area obtained in this paper with the results of other scholars, we found that the law of change is consistent. The studies by Song et al. [40] and Zhang et al. [22] showed that the water area of Baiyangdian Lake reached its maximum in 1989, and the water area from 1989 to 2000 showed a downward trend. From 2000 to 2008, the water area remained unchanged. After 2008, the water area began to rise. This can further indicate that the results of this study are accurate.

Additionally, a more detailed study of the water surface area changes of Baiyangdian Lake is presented in this paper, and the research content is more substantial. The research of Song et al. [40] on the water surface area change in the Xiong'an New Area divides the water surface area change into four stages according to the size of the water area: The lowest level from 1984–1988, the highest level from 1988–1999, the water area from 1999–2006 was less than 40 km^2, and an upward trend from 2007–2016. The research in this paper is divided into four periods according to the water area trend. These periods include the dry lake period of 1984–1988, the declining period of 1989–2000, the constant period of 2000–2008, and the rising period of 2008–2018. In contrast, the phased approach of this paper is more reasonable, and it is easier to explore the reasons for water surface area changes through different period trends. The most important thing is that we divided the water area of Baiyangdian Lake into seven subregions in this paper, and analyzed the variation in each subregion in detail, and the contribution of each subregion to the water area of Baiyangdian Lake.

To date, most researchers [8,40,41] have used the index threshold methods, such as NDWI, for water surface area extractions. In this paper, the SMDPSO method was used to automatically determine the appropriate threshold. This method can avoid the inaccurate extraction result caused by manually selecting the threshold. Moreover, the correlation coefficient between water level and water surface area is 0.96, and most the relative errors of the sample areas are less than 5%.

There are many studies that have highlighted climate change and human activity change as possible causes for the changes of Baiyangdian Lake. Precipitation, temperature, evaporation, and natural inflow were the main analysis factors of climate change. Human activity change also used population and socioeconomic data to be reflected, or only qualitative analysis, due to the lack of direct data [22,38,42]. Moreover, we analyzed the possible causes with four periods separately.

Based on the above analysis, in this study the water surface area extraction accuracy is better, and the water surface area change analysis is more reasonable and detailed.

4.2. Limitations and Prospects

The research in this paper also has certain limitations. The first limitation is that this study does not analyze the seasonal changes in Baiyangdian Lake. The main reason is that the summer water surface cannot be extracted due to vegetation, and the winter water surface extraction is affected by ice and snow. The extraction will not reflect the true seasonality of the water surface. The second limitation is that the analysis of the causes for the change in the water surface of Baiyangdian Lake is not sufficiently deep. There are obvious human activities occurring around and inside Baiyangdian Lake. Moreover, the government has carried out many water diversion projects and human factors have had a great impact on these projects. The complexity of the surrounding environment of Baiyangdian Lake, combined with the common influence of climate and human activities, makes the causal analysis of the water area changes in Baiyangdian Lake very difficult. Our research can only briefly explain the cause of changes in water area through correlation analysis and qualitative analysis.

In future research, determining the cause of the water surface area changes in Baiyangdian Lake is a difficult point. It is important to determine the cause of the decrease in the water surface area. These reasons have an important role in ensuring the ecological environment of Baiyangdian Lake. In addition, remote sensing images from March and April can be used to extract water surface area and avoid the influence of vegetation. In this period, there is no rainy season in northern China, the amount of clouds is small, and the quality of remote sensing images is better.

5. Conclusions

In this paper, the Landsat images from 1984 to 2018 were used to extract the water surface area of Baiyangdian Lake based on the SMDPSO method. The effect of the SMDPSO method on water extraction was verified by visual interpretation and data comparison. The variation law and possible causes of the water surface were also analyzed and discussed. Overall, this study provides the following conclusions:

(1) The SMDPSO method is suitable for the complex water extraction of Baiyangdian Lake. The water extraction results are more accurate in the spring than the other three seasons by accuracy verification, where the relative errors between the observation and the extracted surface area are all less than 5%. The correlation coefficient between water area and water level is 0.96.

(2) The water area of Baiyangdian Lake reached a maximum of 280 km^2 in 1989, and reached a minimum of 44 km^2 in 2002. The change in the water area of Baiyangdian Lake can be divided into four periods: (1) The dry period in 1984–1988, (2) the degraded period in 1989–2000, (3) the stable period in 2000–2008 and (4) the recovery period in 2008–2018. The water surface area of Baiyangdian Lake is mainly concentrated in the five subregions of B, C, D, E, and F. A and G are the main degraded regions. A recovered after 2014, and G had completely degraded into cultivated land after 1999.

(3) The possible causes for the four periods of changes in Baiyangdian Lake are different. The first period of drought and the second period of degradation was caused by climate and human activities. Due to the strengthening of government management and water supply projects, the maintenance of the third period and the recovery of the fourth period were mainly caused by human activities.

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Conflicts of Interest: The authors declare no conflict of interest.

Abbreviations

SMDPSO	Spectrum Matching based on Discrete Particle Swarm Optimization
WL	Water level
AP	Annual precipitation
AAT	Annual average temperature
NI	Natural inflow
WDV	Water diversion volume
WIF	Water inundation frequency

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Water Supply Delivery Failures—A Scenario-Based Approach to Assess Economic Losses and Risk Reduction Options

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Abstract: Access to a reliable water supply is central for a well-functioning society. However, water supply systems are subject to a wide range of threats which may affect their ability to provide water to society. This paper presents a novel risk assessment approach that enables thorough analyses of economic losses and associated uncertainties under a range of water supply disruption scenarios. The purpose is to avoid sub-optimization when prioritizing between risk reduction measures, by integrating the full range of possible outcomes from low to high probability events. By combining risk analysis with cost-benefit analysis, additional information is provided on measures for leveraging investments in managing and reducing the risks. This enables the identification of the most economically profitable risk reduction alternatives and enables decision makers to build strategic capacity for operating in difficult and uncertain futures. The presented approach is exemplified on the island of Gotland, one of the most water scarce areas of Sweden.

Keywords: water scarcity; drought; water supply; risk reduction; risk curves; cost-benefit analysis

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1. Introduction

Water supply infrastructure systems are subject to a wide range of threats which may affect their ability to provide water to society. Predicted population growth and hydro-climatic changes are expected to contribute to both an increased probability of water scarcity and more severe societal consequences [1,2]. In addition to threats related to reduced access to and quality of raw water, failures in water provision may also occur due to events related to the treatment systems, e.g., component failures in treatment plants, and related to the distribution systems, e.g., pipe bursts and pump failures. To deal with the uncertainties and the societal impacts that all these threats entail, risk assessment methods need to be integrated in water supply decision making [3,4]. Risk assessments may be performed in different ways, but a common approach is to qualitatively and/or quantitatively estimate and combine the consequences of one or several possible scenarios, typically undesirable events, the probability of occurrence for the scenarios and the uncertainties related to the included factors [5]. Risk-based decision making uses the results of risk assessments to guide and inform decisions on risk reduction measures. It may, for example, involve comparing required resources for implementing potential risk reduction measures with potential benefits of estimated risk reduction. A framing based on risk provides for a better understanding of the severity, distribution and impacts of the full range of possible outcomes [6].

Decision-makers and water supply managers face difficult decisions on resource allocation and prioritizations of risk reduction measures. To support such decisions, effective risk management requires the identification and assessment of a range of representative risk scenarios [7]. The process of summing and showing the interaction between single or individual risks is sometimes referred to as risk aggregation [8]. Moreover, to facilitate rational decision-making, the risk should be expressed in a clear manner, and related uncertainties considered [9]. In this paper, the focus is on economic consequences, and risk is expressed in terms of economic consequences to society arising from water supply disruption events. A disruption in the water provision can lead to economic consequences for the water utility as well as for businesses and residential consumers, and may generate significant economic losses for society [10]. Several different methods have been used to estimate business interruption losses, e.g., input-output models and computable general equilibrium models [11]. The direct economic consequences to commercial and industrial consumers are often estimated by use of importance (or resiliency) factors, i.e., quantitative measurements focusing on the production output during disruption [12]. Residential welfare loss of water supply disruptions can be assessed based on estimates of consumer willingness to pay to avoid such disruptions [13,14]. Short-term disruption events are not evaluated as frequently as long-term disruptions. They may, however, contribute significantly to the total economic losses due to their much higher frequency [10].

According to Uzielli, et al. [15], a quantitative risk assessment should include a quantification of the expected losses, based on the probability for a given event, the economic consequences to society of exposed elements at risk, and their associated vulnerability. However, risk assessments are often complex in nature, and many aspects of the risk may be subject to large uncertainties [16]. It is now common to define risk using uncertainty as a key factor, see e.g., International Organization for Standardization (ISO) [17] and Aven [18]. The importance of considering uncertainties is particularly true for factors affecting high-impact-low-probability risks which, by their very nature, occur only infrequently. Existing statistics may be insufficient to support the risk assessments [19]. Data samples may, for example, be too small, too unreliable, too costly to obtain, or simply unobtainable. In these cases, the only sound option may be to elicit the information needed using expert judgements. The typical way is to elicit judgement from more than one expert and represent the uncertainties by probability distributions [20–22], so that appropriate decisions can be made on risk reduction. The approach proposed in this paper integrates the full range of risk scenarios, while taking the underlying uncertainties into account, to estimate the total risk of the water supply system. This allows for a better understanding of how different factors influence each component of risk and how they, in turn, affect the total risk. It further facilitates a design and prioritization of measures that focus on addressing the total risk rather than individual threats.

The overall aim of this paper is to provide a risk assessment method that enables thorough analyses of risk reduction measures by integrating the full range of possible outcomes from low to high probability events. The purpose of the method is to provide a structured and thorough analysis of the total risk to enable prioritization of possible measures based on, e.g., economic profitability. A key part is also to avoid sub-optimization, where risk reduction measures are prioritized based on individual events. Specific objectives are to: (1) provide a method that enables estimation of economic losses under various levels of water supply disruption events; (2) combine this information with the integrated likelihood function of disruption events to estimate the total risk under existing conditions; (3) analyze and compare the annual benefits and economic profitability of risk reduction measures; and (4) exemplify this method by application on the island of Gotland, Sweden. The proposed approach is a valuable contribution to the water supply reliability literature, in which definition of risk scenarios, uncertainty estimations of input variables, economic valuation of consequences, calculations of the total integral sum of risk over different risk scenarios and calculations of economic profitability through cost benefit analysis (CBA) all are rare.

2. Materials and Methods

In short, the risk assessment method described in this paper is based on a combination of quantitative risk analysis and CBA [23]. The main steps are: (1) identification of risk scenarios;

(2) estimation of factors affecting the risk; (3) characterization of risk; (4) evaluation of risk reduction measures; and (5) performance of uncertainty and sensitivity analyses. The methodology is described in more detail in the following paragraphs.

2.1. Identification of Risk Scenarios

In this paper, a well-established approach to risk analysis is used where the aim is to answer the following three questions [9]:

1. What can go wrong?
2. How likely is it to happen?
3. If it does happen, what are the consequences?

To answer these questions, a set of scenarios are defined. The set of scenarios used in a quantitative risk analysis should preferably be complete, finite and disjoint [24]. This means that a nonoverlapping subset of N scenarios together should represent all possible risk scenarios for the entire problem so that the total risk R is

$$R = \{(s_i, f_i, x_i)\} \quad (1)$$

where s_i is scenario i, $i = 1, 2, \ldots, N$; f_i is the frequency with which the scenario occurs; and x_i is the consequence given that scenario i occurs. Furthermore, the uncertainties related to the three variables are identified and described quantitatively or qualitatively to enable a thorough description of the risk.

There are several scenario identification methods used within the theory of scenario structuring (TSS), e.g., hierarchical holographic modeling (HHM), failure modes and effects analysis (FMEA), hazard and operations analysis (HAZOP), and anticipatory failure determination (AFD). All methods start by defining a success scenario. The risk scenarios (s_i) can then be identified by decomposing the success scenario into different parts, e.g., in geographical, hydrological, temporal or functional parts, and asking; "What can go wrong in this part?" or "What happens if this parameter changes?". The aggregated risks of all scenarios then determine the total risk of the overall system [24].

2.2. Estimation of Factors Affecting Risk

As mentioned above, the risk is defined as a function of a set of scenarios, the frequency with which they occur and the consequences if they occur. When we do not know the frequencies or the consequences with certainty, we can express them by probability distributions so that $R = \{(s_i, p_i(f_i), \zeta_i(x_i))\}$, where p_i and ζ_i are the probability density functions for the frequency and consequence, respectively. In this study, the following economic consequences are considered: residential welfare losses, businesses losses, and water utility expenditures for upholding water provision (as far as possible) during the disruptions. The water utility expenditures were estimated based on information from the local water utility from previous experiences. In the subsections below it is described how uncertain quantities, of e.g., the return periods and duration of events, were estimated based on formal expert elicitation, and how residential welfare losses and business losses were calculated.

2.2.1. Formal Expert Elicitation

One technique of capturing the probability distributions of uncertain quantities is to elicit this information using a range of experts from different disciplines. In this paper, uncertain quantities, such as return period and duration of events, were estimated by expert elicitation using the Sheffield Elicitation Framework (SHELF) [25]. The SHELF framework elicits a single judicious consensus distribution from the expert group for each uncertain quantity. The process begins by eliciting individual judgements from each expert independently, followed by a group discussion and a group judgement. The parameters estimated in this paper were the lower and upper plausible limits for the uncertain quantity, as well as the median and lower and upper quartiles. The MATCH Uncertainty

Elicitation Tool [26] was used to find the best fitted statistical distribution model for the group judgment and to provide direct visual feedback to the expert. The results were reviewed and discussed by the group and when necessary adjusted to fit their final and joint preferences.

2.2.2. Estimation of Household Welfare Losses

One consequence of water disruptions is residential welfare losses. In this paper, this was valued based on estimates of consumer willingness to pay to avoid water supply shortages [13,14]. By integrating the demand curve for water, between baseline consumption and reduced consumption, the daily welfare loss $W_i(z_{jt})$ for a consumer in region j facing a water shortage of z at time t was calculated as:

$$W_j(z_{jt}) = \frac{\eta}{1+\eta} Y_{baseline} Q_{baseline} \left[1 - \left(\frac{Q_r(z_{jt})}{Q_{baseline}} \right)^{\frac{1+\eta}{\eta}} \right] \quad (2)$$

where $Y_{baseline}$ is the average water price when no shortage, $Q_{baseline}$ is the average amount of water consumed per capita per day when no shortage, Q_r is the reduced water consumption, and η is the price elasticity of water demand. The severity of the water shortage was defined as z_{jt} ϵ [0,1], where $z_{jt} = 0$ corresponds to no water and $z_{jt} = 1$ corresponds to normal water availability [13].

The average water price on Gotland in 2017 was 12.74 SEK/m^3 (100 Swedish Krona (SEK) ≈ 10 USD in October 2019) and the average amount of water consumed was 132 L per capita and per day [27]. There is no price elasticity estimate available for Gotland. Therefore, a mean price elasticity of water demand for developed countries (−0.378) was applied, based on the meta-analysis by Sebri [28] (p. 518). For sensitivity analysis, a price elasticity of −0.2 was used, following a study of household water demand in Sweden [29]. Q_r was estimated at the SHELF workshops.

2.2.3. Estimation of Business Losses

Another consequence following water disruptions is the economic consequences for commercial and industrial customers due to loss of potable water service. In this paper, the estimation of value added lost for businesses followed the Federal Emergency Management Agency (FEMA) [30] (p. 39) methodology of using local GDP data [31] in combination with water importance factors [32]. It was here assumed that Swedish economic sectors have the same percentage reduction in value added from water supply disruptions as US economic sectors.

2.3. Risk Characterization

A risk curve for the reference alternative, i.e., the current water supply system, is developed based on the triplets (s_i, f_i, x_i). For this, the scenarios must first be arranged in order of increasing consequences, i.e., $x_1 \leq x_2 \leq \ldots x_i \leq \ldots \leq x_N$, along with corresponding frequencies. Starting with the scenario with the most severe consequences, a cumulative frequency F_i, i.e., the frequency of having consequence equal to or greater than x_i, is calculated as $F_i = F_{i+1} + f_i$. By plotting (x_i, F_i), a staircase function of the analyzed risk scenarios is derived, representing a discrete approximation of the continuous reality. A smoothed risk curve R_x, drawn through the staircase (Figure 1), can then be regarded to represent the actual risk [9]. Each point of the curve does not belong to a specific event but instead represents the estimated return period of losses. The integral of the curve, i.e., the area underneath the curve, represents the total expected losses in any given year so that:

$$R_{tot} = \int_0^{x_N} F(x)dx \quad (3)$$

where R_{tot} is the total annual risk, N is the total number of analyzed scenarios, x is the combined economic consequences for the municipality, households and businesses (i.e., $x = x_{Municipality} +$

$x_{Households} + x_{Bu\,\sin esses}$), and F is the cumulative frequency as a function of consequence x. For risk estimation, the continuous function is simplified by the staircase function.

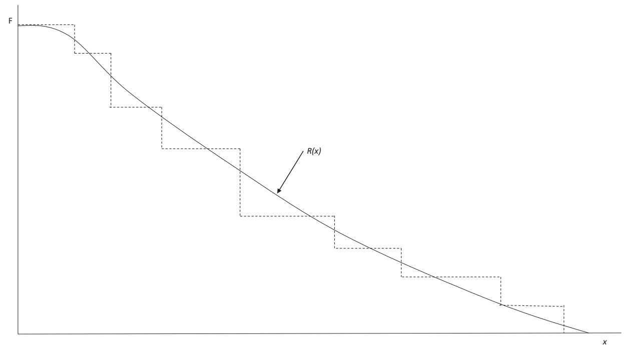


Figure 1. Schematic description of staircase and continuous risk functions based on e.g., Kaplan and Garrick [9].

2.4. Evaluation of Risk Reduction Measures

A risk reduction measure is here defined as any measure that can be applied to reduce the frequency and/or the consequences of the undesirable events. The same scenarios (s_i) used when estimating the risk level of the reference alternative are also used to assess potential risk reduction measures, but the measures' associated frequencies and consequences are applied. For each measure (a), a new risk curve is created and thus a new annual total risk. The annual risk reduction, i.e., the annual benefit B_a, is calculated as the difference between the risk curve of the reference alternative R_0 and the risk curve of the analyzed measure R_a as $B_a = R_0 - R_a$.

To compare the economic profitability of implementing the measures, a CBA [23] was performed. CBA is a structured method to compare the societal costs of an option with its benefits. The estimated risk reductions were included in the CBA as annual benefits [33]. The decision-metric of the CBA is the net present value (NPV), calculated as:

$$NPV_a = \sum_{t=0}^{T} \frac{B_{a,t} - C_{a,t}}{(1+r)^t} \tag{4}$$

where a is the alternative measure, t is the time when benefit or cost occur, T is the time horizon, r is the discount rate, C are the costs associated with implementing a risk reduction measure, and B is the benefit of risk reduction in relation to the reference alternative. A measure is considered economically profitable when its total benefits to society are larger than its total costs to society, i.e., when its NPV is positive. Three discount rates were used (1.4%, 3.5% and 5%, respectively), reflecting the average discount rate used in the Stern Review on Climate Change [34] and the suggested social and private rates of the Swedish Transportation Administration Guidelines for cost-benefit analysis [35].

2.5. Uncertainty and Sensitivity Analyses

This paper applies a probabilistic approach with formal uncertainty analysis. As described above, the SHELF Framework was used to elicit information regarding uncertain quantities such as the proportion of households affected in different scenarios and the frequency of events. Probability distributions were assigned to represent each uncertain quantity, and Monte Carlo simulations (10,000 iterations) were used to calculate the annualized risks, risk reductions and NPVs using the risk analysis software @Risk 7.6.0 (Palisade, Ithaca, USA). This provides important additional information for the decision-makers. As Kaplan and Garrick [9] (p. 14) state, a single number is not a big enough concept to communicate a risk—it takes a whole family of risk curves. The uncertainties in input data can, for example, be used to visualize the resulting mean, minimum and maximum risk curves. The uncertainties can also be transferred in loss exceedance curves, i.e., the probability that the expected loss exceeds a certain value [36]. Since it is hard, and often not possible, to capture all uncertainties in the variables of a risk model, other uncertainty factors are identified, described and discussed using a qualitative approach. The purpose is to provide a transparent decision support that highlights uncertainties that may affect the interpretation of the results.

3. Method Application

3.1. The Case Study Site

The case study site was the island of Gotland (3000 km^2) in Sweden, located in the Baltic Sea about 100 km east of the mainland and with a population of 58,000. Gotland suffers from low water availability and difficulties in providing enough water to the society. The island's thin soil layers, lack of coherent reservoirs in the limestone bedrock and extensive drainage of arable land, result in an overall low storage capacity of water and a high precipitation run-off [37]. Climate change is expected to further limit the water availability on the island. Longer dry periods are predicted during summers, and the groundwater recharge is expected to decrease due to an increased temperature and the subsequent increase in evaporation and vegetation periods. Currently, about 18 million cubic meter per year is used by households (4 Mm^3), animal keeping (1.5 Mm^3), tourism (1.3 Mm^3), industry (6.1 Mm^3) and irrigation (5 Mm^3) [38]. A large proportion of the water supply is based on private solutions. For example, only 67% of the households are connected to the public water supply system, which during the summer months to 40% is based on groundwater, 20% on surface water and 40% on desalinated seawater [39].

Gotland is one of the most popular tourist summer destinations in Sweden. In 2016, over 2 million people traveled to Gotland, and the number of guest nights at hotels and other commercial accommodation facilities exceeded 1 million [40]. Hence, there is a large seasonal variation of water demand on the island with the highest demand occurring when the water supplies are at their lowest. In addition to an already constrained water supply situation, the total water demand, i.e., of municipal water provision and other water sources, is expected to increase by about 40% by 2045 with increases of 30% in tourism, 20% in domestic demand, 20% in animal keeping, 15% in industry, and 100% in irrigation [38]. The current water resources on the island cannot meet this projected increase in demand, especially during the summer months. Due to Gotland's insular location there is also no possibility to strengthen the water supply from neighboring municipalities.

3.2. Scenarios and Risk Reduction Measures

Six scenarios were identified around the question: What can pose a challenge to maintain a continuous municipal water supply provision on Gotland? see Table 1. The scenarios were developed during multiple discussions with the municipality's water supply strategists to represent the range of possible events that may present challenges to the municipal water supply. More detailed information about the scenarios was discussed at the workshops but is confidential for safety reasons.

Based on previous estimates of where, and with how much, the municipality can increase groundwater and surface water abstractions as well as supplement groundwater catchments by managed aquifer recharge (MAR) [37,41], four alternative risk reduction measures were analyzed in this paper, see Table 2. Focus in this paper is hence on improvements in the raw water system. The analyzed measures are site specific, thus they can reduce the risk in the areas in which they are applied but not in areas to which, e.g., the distribution network is not connected. The reason desalination is not further explored is because the municipality has decided to prioritize freshwater (from groundwater, lakes and streams) over seawater for public water supply. Desalination is to be further considered only if the freshwater resources cannot meet demand [42].

Table 1. Scenario summaries.

Scenario	Summary
Scenario 1	One of the smaller towns (with approximately 400 inhabitants) experiences failure in the water supply provision. This can be caused by failures in either the distribution system, the raw water system or the treatment system. The municipality transports water by truck to the town.
Scenario 2	The water availability on the small, adjacent island of Fårö is too low during summers to meet demand. The municipality transports water to the island. The amount of water trucked varies over the summer months with the number of tourists on the island.
Scenario 3	Due to low precipitation, the raw water quantity is insufficient approaching the summer months. The municipality prohibits urban irrigation and call for careful use of the drinking water.
Scenario 4	A failure in connection to the municipality's desalination plant makes it unable to provide water to consumers. The nearby groundwater resource is used as a backup. The amount of available groundwater is, however, not sufficient, and households, summer tourists and businesses in that region have to make do with a reduced water quantity.
Scenario 5	One of the larger towns (with approximately 1500 inhabitants) experiences failure in the water supply provision. Again, this can be caused by failures in either the distribution system, the raw water system or the treatment system. The municipality transports as much water as possible to the town, but households and businesses in that town must make do with a reduced water quantity.
Scenario 6	Due to a severe drought, neither the groundwater nor the surface water resources are sufficiently replenished. Households and businesses on the whole of Gotland have to make do with a significantly reduced water quantity.

Table 2. Alternative risk reduction measures.

Measure	Summary
MAR	Managed aquifer recharge (MAR) in nine of the municipality's existing well fields. In total, an additional 490,000 m^3 is made available annually.
GW	Increased groundwater extraction (GW) from three groundwater resources on Gotland. In total, an additional 2 million m^3 is made available annually.
SW small	Increased surface water extraction (SW small) from one of the surface water resources on the island. In total, an additional 380,000 m^3 is made available annually.
SW large	Increased surface water extraction (SW large) from one of the surface water resources on the island. In total, an additional 4.7 million m^3 is made available annually.

To estimate uncertain factors affecting identified scenarios and risk reduction measures, three half day SHELF elicitation workshops [25] were held in May and June 2019, with workshop participants ranging from 2 to 6 experts and 1 to 3 workshop facilitators. The workshop participants (6 in total) represented the following areas of expertise: public drinking water management, public water supply strategy, emergency management, environmental expertise, private water supply, and longtime operational water utility staff. For the more frequent events, there was plenty of background information to rely on regarding, e.g., estimation of different cost aspects. For the more infrequent events, the estimations were naturally more speculative.

4. Results

Details on quantified variables from the SHELF workshops and follow up meetings are provided in Table 3. The table provides the input variables for the calculations of total risk, risk reduction and net present values (NPV), performed by Monte Carlo simulations. For a few events that are expected to occur each year, uncertainties regarding frequency and return period were not quantified. Frequency was generally used as a measurement of occurrence when the estimated time between events was greater than one time per year; otherwise the return period was used.

Table 3. Estimates of input variables. Distributions used are log-normal, LN (μ; σ), beta, B (α1; α2; min; max) and gamma, G (α; β). R0 = reference alternative; GW = groundwater extraction; MAR = managed aquifer recharge; SW small = surface water extraction small scale; SW large = surface water extraction large scale.

Input Variable (Scenario)	Unit	R0	GW	MAR	SW Small	SW Large
Frequency (1)	1/year	G (6.4; 0.86)	B (0.92; 2.3; 1; 10)	G (6.4; 0.86)	B (0.57; 1.4; 1; 7)	B (0.57; 1.4; 1;7)
Duration (1)	days	LN (1.7; 0.53)	LN (1.7; 0.53)	LN (1.7; 0.53)	LN (1.7; 0.53)	LN (1.7; 0.53)
Transportation (1)	SEK/day	12,516	12,516	12,516	12,516	12,516
Frequency (2)	1/year	1	1	1	1	1
Duration (2)	days	B (1.0; 0.98; 30; 60)	B (1.0; 0.98; 30; 60)	B (1.0; 0.98; 30; 60)	B (1.0; 0.98; 30; 60)	B (1.0; 0.98; 30; 60)
Transportation (2)	SEK/day	8344	8344	8344	8344	8344
Return period (3)	years	1	B (0.75; 1.6; 1; 3)	B (0.75; 1.6; 1; 3)	1	B (0.43; 2.6; 10; 500)
Duration (3)	weeks	B (1.3; 1.0; 3; 25)	B (1.0; 2.0; 3; 25)	B (1.0; 2.0; 3; 25)	B (1.3; 1.0; 3; 25)	B (1.3; 1.0; 3; 25)
Reduced water consumption due to prohibited irrigation (3)	%	B (3.8; 2.1; 3; 13)	B (3.8; 2.1; 3; 13)	B (3.8; 2.1; 3; 13)	B (3.8; 2.1; 3; 13)	B (3.8; 2.1; 3; 13)
People affected by prohibition (3)	#	37,250	37,250	37,250	37,250	37,250
Reduced water consumption due to information on careful water use (3)	%	B (2.8; 2.2; 2; 7)	B (2.8; 2.2; 2; 7)	B (2.8; 2.2; 2; 7)	B (2.8; 2.2; 2; 7)	B (2.8; 2.2; 2;7)
People affected by information (3)	#	59,250	59,250	59,250	59,250	59,250
Return period (4)	years	B (0.92; 2.3; 1; 10)	B (0.92; 2.3; 1; 10)	B (0.92; 2.3; 1; 10)	B (0.92; 2.3; 1; 10)	B (0.92; 2.3; 1; 10)
Duration (4)	days	B (1.4; 21; 1; 270)	B (1.4; 21; 1; 270)	B (1.4; 21; 1; 270)	B (1.4; 21; 1; 270)	B (1.4; 21; 1; 270)
People affected by reduced consumption (4)	#	25,000	25,000	24,200	25,000	25,000
Economic impact for people and businesses of reduced water consumption (4)	SEK/day	2,812,300	2,812,300	2,736,700	2,812,300	2,812,300
Return period (5)	years	B (1.1; 5.6; 3; 20)	B (1.5; 7.9; 4; 30)	B (1.1; 5.6; 3; 20)	B (1.5; 7.9; 4; 30)	B (1.3; 6.5; 5; 35)
Duration (5)	days	B (0.89; 13; 3; 600)	B (0.89; 13; 3; 600)	B (0.89; 13; 3; 600)	B (0.89; 13; 3; 600)	B (0.89; 13; 3; 600)
Reduced water consumption due to lowered water pressure (5)	%	B (5.8; 5.8; 4; 6)	B (5.8; 5.8; 4; 6)	B (5.8; 5.8; 4; 6)	B (5.8;5.8;4;6)	B (5.8;5.8;4;6)

Table 3. *Cont.*

Input Variable (Scenario)	Unit	R0	GW	MAR	SW Small	SW Large
People affected by lowered pressure (5)	#	1500	1500	1500	1500	1500
Transportation (5)	SEK/day	41,100	41,100	41,100	41,100	41,100
Return period (6)	years	B (1.0; 1.0; 30; 200)	B (1.0; 1.0; 30; 200)	B (1.0; 1.0; 30; 200)	B (1.0; 1.0; 30; 200)	B (1.0; 1.0; 30; 200)
Duration (6)	days	B (0.71; 0.64; 30; 80)	B (0.71; 0.64; 30; 80)	B (0.71; 0.64; 30; 80)	B (0.71; 0.64; 30; 80)	B (0.71; 0.64; 30; 80)
Reduced water consumption for people and businesses (6)	%	50	50	50	50	23,7
People affected by reduced consumption (6)	#	96,750	96,750	96,750	96,750	96,750
Reduced county GDP (6)	million SEK/day	12.7	12.7	12.7	12.7	4.2
Cost of piping and wells etc. (mean values)	million SEK		18	45		8
Costs of construction and treatment components * (mean values)	million SEK		16.5	9	15	49

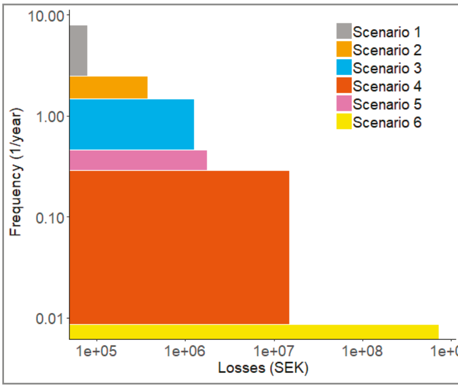
* Some treatment component costs are recurring every 7 or 10 years [41].

The results and related uncertainties are dependent on the estimated input variables but also on the basic assumptions used to describe the system and the future development. Table 4 provides information on non-quantified uncertainty factors discussed at the SHELF workshops, along with the associated assumptions made. This qualitative analysis of uncertainties is of great importance when interpreting the results.

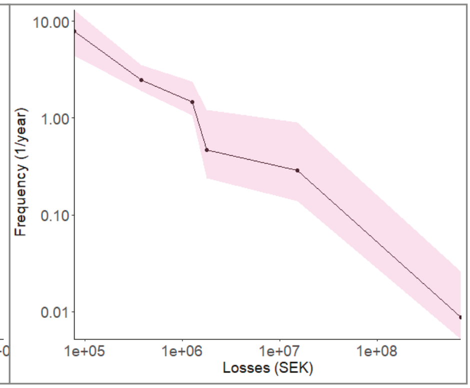
Table 4. Non-quantified uncertainty factors discussed at the Sheffield Elicitation Framework (SHELF) workshops.

Uncertainty Factor	Description
Effect of information over time	The residential water consumption was estimated to decrease by about 5% when the municipality calls for careful use of drinking water. It is uncertain how effective such information is over time and, hence, if the effect is maintained over the summer months. The effect might also decrease from one year to another because a larger portion of households have invested in residential water saving technologies. It was here assumed that the effect stayed the same over time.
Temperature	The residential water consumption varies with outside summer temperature. The high summer temperatures of 2018, for example, resulted in people showering more than normal, which increased the water consumption. The effect of varied summer temperatures was not taken into account. Residential water consumption was instead based on the daily average consumption on Gotland.
Precipitation	The residential water consumption was estimated to decrease by about 10% when the municipality prohibits urban irrigation. The respect for such prohibitions tends to decrease if/when it rains, and the effect may hence vary over time. It was here assumed that the effect stayed the same over time.
Geographical spread	Households tend to be more inclined to decrease water consumption when the water shortage has national implications, partly due to the larger media focus of national compared to local water shortages. The geographical spread also affects the possibility of getting help from other municipalities, e.g., in the form of trucks for water transportation. It was here assumed that, at least, the southern part of Sweden experienced water shortage at the same time as Gotland.
Tourism	It is uncertain how an extreme drought will affect tourism, and whether tourists will travel to Gotland to the same extent as usual. It was here assumed that the number of tourists on Gotland was not affected by water shortages or extreme droughts.

The estimated annual risk for the reference alternative R_0 is demonstrated in Figure 2 in the form of a staircase to the left and as a risk curve showing the mean and P05 and P95 frequency percentiles to the right. According to calculation results, the low-frequency events are generally associated with larger economic consequences than high-frequency events. However, the annual risk is the lowest for the second least frequent event (Scenario 5) and the highest for the most frequent event (Scenario 6): 425,000 SEK for Scenario 1; 378,000 SEK for Scenario 2; 1,262,000 SEK for Scenario 3; 4,222,000 SEK for Scenario 4; 309,000 SEK for Scenario 5; and 6,321,000 SEK for Scenario 6 (mean values). The total annual risk is estimated at approximately 12,916,000 SEK, ranging from 7,161,000 SEK to 32,370,000 SEK for the 5th and 95th percentiles, respectively (mean values).



(a)



(b)

Figure 2. Estimated annual risk of the reference alternative for analyzed scenarios in the form of staircase (**a**), and in the form of a risk curve showing the mean values and frequency percentiles P05 and P95 (**b**). Note that the curves are plotted on log-log scales with cumulative frequencies.

The risk curves of the alternative risk reduction measures are shown in Figure 3 along with the risk curve of the reference alternative. The potential risk reduction of the measures is the difference between the risk curve of the reference alternative and those of the measures. The large-scale surface water measure was shown to reduce the total annual risk the most, suggesting a potential reduction of approximately 6 million SEK annually compared to 965,000 SEK for groundwater, 785,000 SEK for MAR, and 307,000 SEK for the small surface water measure (mean values).

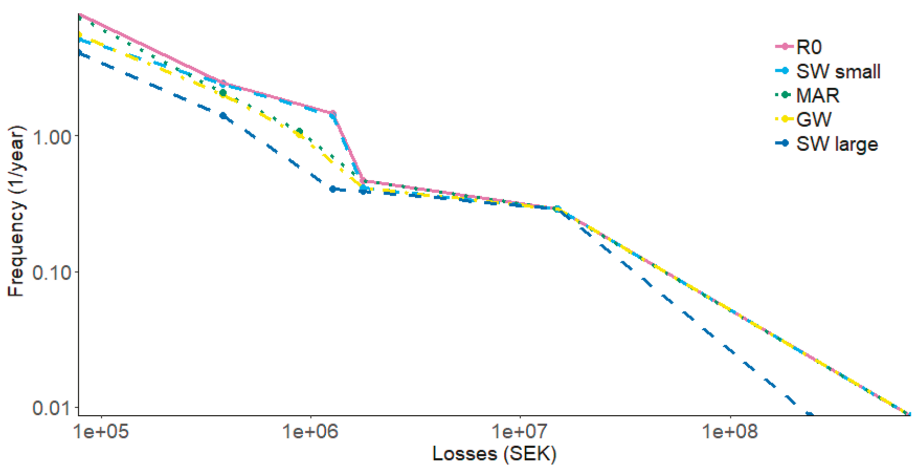


Figure 3. Risk curves for analyzed risk reduction measures over all scenarios (mean values). Note that the curves are plotted on log-log scale with cumulative frequencies.

The probabilities of each measure being the best option with respect to risk reduction for each individual scenario and combined for all scenarios is shown in Figure 4. The results show that the large-scale surface water measure has the highest probability to be the best option for most individual scenarios and for all scenarios combined. The ranking order of the other measures vary between risk scenarios.

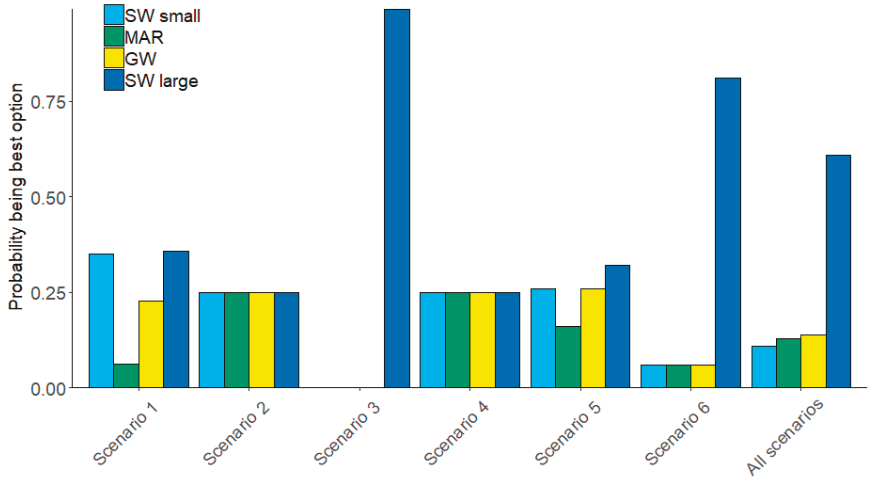


Figure 4. Probability that each measure is the best option with respect to risk reduction.

The result from the cost-benefit analysis is shown in Figure 5 displaying that the large-scale surface water measure was the least economically beneficial measure for Scenarios 1 to 5 when analyzed individually, but the most beneficial measure for Scenario 6 and when including the risk reduction for all scenarios combined. The ranking order of the other measures varied somewhat between the analyzed scenarios. The NPV mean values in million SEK for the measures SW small, MAR, GW, and SW large respectively are: −37, −54, −46, and −108 (Scenario 1); −42, −54, −50, and −113 (Scenario 2); −42, −38, −33, and −83 (Scenario 3); −42, −53, −50, and −113 (Scenario 4); −40, −54, −47, and −109 (Scenario 5); −42, −54, −50, and −13 (Scenario 6); and −35, −36 −27 and 24 (all scenarios combined). It is worth noting that the NPVs are based only on implementation costs and the benefits of risk reduction with respect to water supply disruptions. The CBA could therefore be improved by inclusion of other costs and benefits, e.g., relevant ancillary effects. However, the present result is sufficient to highlight the importance of a holistic view based on multiple scenarios when prioritizing between risk reduction measures.

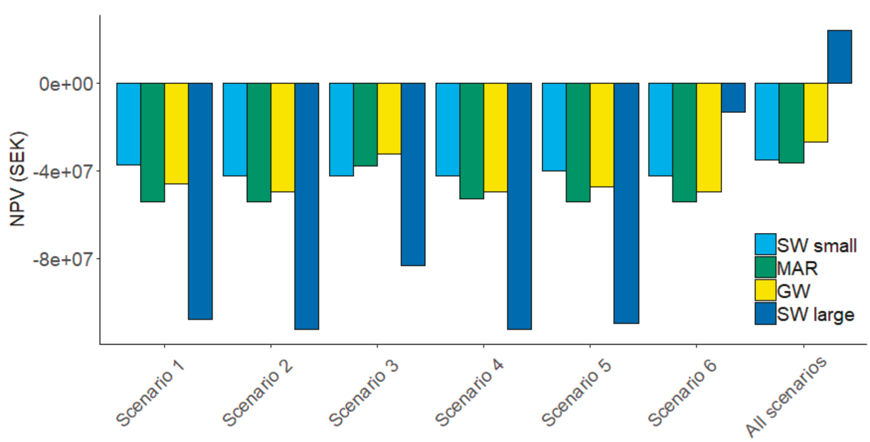


Figure 5. Net present values for measure implementation with the annual benefit of risk reduction for each individual risk scenario and for all scenarios combined, over a 50-year time horizon and with 3.5% discount rate (mean values).

The economic benefit of risk reduction is distributed differently across households, businesses and the municipality for the analyzed measures and scenarios. In Scenario 1, the municipality gained 100% of the benefits. In Scenario 2, no measure contributed with any benefit of risk reduction. In Scenario 3, the households gained 100% of the benefits. In Scenario 4, the households gained 100% of the benefits of MAR. The other measures did not contribute to any benefits in that scenario. In Scenario 5, the municipality gained 99.7% of the benefits of the increased groundwater extraction and the small- and large-scale surface water measures, and the households gained 0.3% of the benefits. In Scenario 6, the businesses gained 99.2% of the benefits of the large-scale surface water measure and the households gained 0.8%. No other measure contributed with risk reduction in that scenario.

Results from the two forms of sensitivity analyses performed (based on scenario analysis and Monte Carlo simulations respectively) are provided in Table 5 and Figure 6. Table 5 shows that the ranking order of the measures did not vary much when applying different discount rates. However, the order of the measures varied when applying different price elasticities. Particularly the MAR measure benefited from the −0.2-price elasticity compared to the other measures.

Table 5. Ranking order of net present values for analyzed measures when using two different price elasticities of water demand and three different discount rates. Rank 1 = highest net present value (NPV) and Rank 4 = lowest NPV (mean values). The risk reduction of all scenarios combined are used in these calculations.

Price Elasticity	**−0.378**			**−0.2**		
Discount rate	1.4%	3.5%	5%	1.4%	3.5%	5%
SW small	4	3	3	4	4	4
MAR	3	4	4	2	2	2
GW	2	2	2	3	3	3
SW large	1	1	1	1	1	1

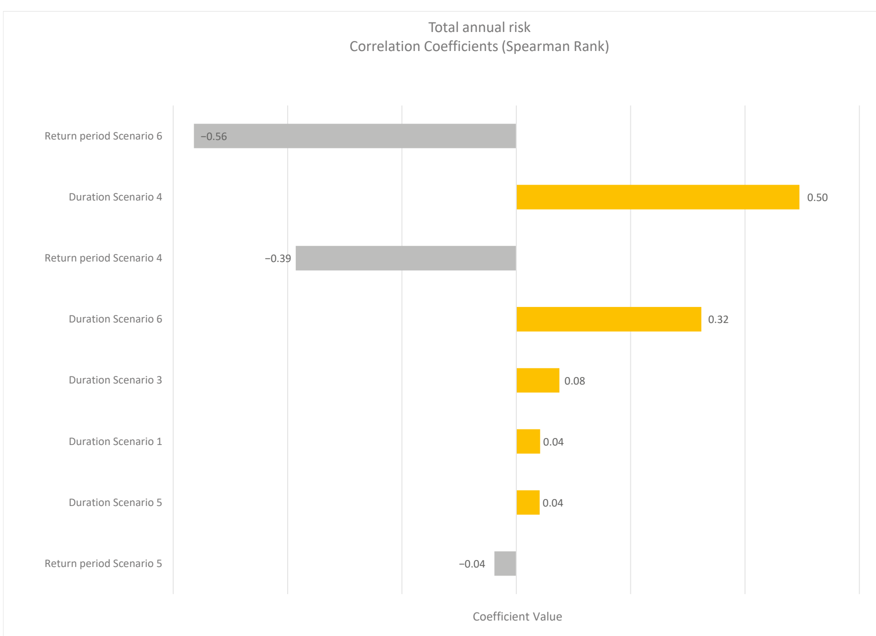


Figure 6. Correlation coefficients (Spearman rank) of the eight most strongly correlated input variables for the total annual risk.

Figure 6 shows the degree to which input variables co-vary with the calculated total risk, expressed using Spearman rank correlation coefficients between −1 and 1. Input variables related to the return periods and duration of the risk scenarios contributed more to the outcome uncertainty than input parameters related to the economic consequences of the scenarios. This holds true also when comparing how the input variables co-vary with the estimated NPVs, i.e., input variables related to return periods and duration of risk scenarios contributed most to the NPV uncertainties.

5. Discussion

Gotland's drinking water system is vulnerable to supply and demand fluctuations. Insufficient water availability in combination with rainfall deficiencies and large seasonal demand variations pose challenges to the local water utility. In addition, the total water demand on the island is expected to increase by more than 40% over the coming decades [38]. Taken together, the low and varied water availability coupled with other threats to the drinking water system, illustrates the importance of understanding the system risks as well as the benefits of investing in a reliable water supply [14]. In this paper, four potential risk reduction measures were analyzed for Gotland, providing guidance on how efficient the measures are to reduce different types of risks. The risk analysis was combined with cost-benefit analysis to provide information on the measures' economic viability. For Gotland, the large-scale surface water measure (SW large) proved to be the most beneficial measure for reducing the risk in most individual risk scenarios and in all scenarios combined. However, the large-scale surface water measure was the least economically beneficial measure for the individual Scenarios 1 to 5 when comparing NPVs, but the most economically beneficial measure for Scenario 6 and when including the risk reduction of all scenarios combined. This is because the measure has high implementation costs but also a high risk reduction effect on several of the scenarios, and the combined effect of these risk reductions creates a large benefit. The varying ranking order of the measures for Gotland, when analyzing risk reductions for individual scenarios versus all scenarios combined, highlights the importance of a holistic risk assessment, integrating a range of risk scenarios. This is to avoid sub-optimization where measures are prioritized based on individual risk scenarios. By calculating the total risk, the possibility of more than one scenario occurring simultaneously is considered. However, it should be noted that the measures analyzed in this paper focused mostly on improving the raw water system, and little attention was given to improving the treatment system or the distribution system.

The presented method makes use of a non-overlapping subset of risk scenarios, which together should represent all possible scenarios for water supply disruptions. By quantifying the probability of losses caused by the scenarios, a risk curve is produced showing the relationship between frequency and its associated losses. Each point of the curve represents the actual return period of losses, and the curve can hence be used to provide information on how to address the different levels of risk. In the paper, we have chosen to express the risk in terms of expected economic consequences to society arising from disruption events. However, it is important to point out that in other situations there may be reasons to express the risk in other terms, in which case the same method can still be used. It is also important to note that there are limitations in expressing the risk in terms of expected consequences, particularly when it comes to capture events with low probabilities and high consequences [18]. However, we have judged the type of events we consider are the type that can be assessed with expected consequences.

Rational decision-making requires that the risks, along with other costs and benefits, are properly accounted for in the decision-making process [9]. However, evaluations of alternative measures and their effects will always comprise uncertainties. In this paper, the uncertainties of input variables were represented by probability distributions, and the uncertainty of the outcomes were calculated by means of Monte Carlo simulations. This approach allows us to study the uncertainty in the results and the likelihood of each outcome. It also facilitates sensitivity analysis, e.g., using Spearman rank correlation coefficients, to study how uncertainties of specific input variables contribute to the uncertainties in the results. Such information can for example be used to support decisions on which input variables to prioritize for further research and/or data collection in order to reduce uncertainties in results. However, it is practically impossible to cover all aspects of real systems [24]. Hence, the assigned probabilities are conditioned on several assumptions and simplifications. For assumptions and simplifications not to be overlooked in the risk management and decision-making processes, these variables are included in the analysis using a qualitative approach as suggested by e.g., Aven [18] (p. 630). This approach highlights basic assumptions that, for example, affect the estimated input variables. If the analysis would have been based on another understating of the system and its development, the results would of course have been different. Hence, the qualitative uncertainty analysis provides transparency and is of great importance when interpreting the result. Further, some discrete uncertainties, such as discount rates and price elasticities, are analyzed by use of scenario analysis [43]. This comprehensive handling of uncertainties demonstrates a structured and transparent way of expressing risk so that water utilities can use estimates of failure rates and welfare losses over a range of disruption scenarios to identify the measures that will lead to the lowest economic losses for society, and hence improve water supply planning and risk management.

It should be emphasized that despite the abundant information provided by the risk assessment approach, its most important contribution may be that it initiates a process in which aspects otherwise likely overlooked or ignored are openly addressed. For example, definition of risk scenarios, uncertainty estimations of input variables, economic valuation of consequences, calculations of the total integral sum of risk over different risk scenarios, and calculations of economic profitability through CBA are all rare in water supply reliability studies. However, we did not consider combinations of risk reduction measures or effects on other externalities, such as health issues or agricultural production. Hence, the provided method can be improved by enabling assessments of measure combinations and inclusion of other relevant costs and benefits.

6. Conclusions

The main conclusions of this paper are:

- The risk-based approach proposed in this paper can be used to evaluate uncertainties and provide information on frequencies and welfare losses of water supply disruptions. By evaluating a range of scenarios, decision makers become aware of the strengths and weaknesses of their water supply system. An increased knowledge of the risks allows for an understanding of how to address the threats and can be used as a starting point for identifying risk reduction measures. The approach enables decision makers to build strategic capacity for operating in difficult and uncertain futures.
- In the proposed approach, alternative measures can be evaluated and compared based on their risk reduction capacities, highlighting whether they reduce the frequencies and/or the consequences of identified risk scenarios. By combining the risk analysis with cost-benefit analysis, additional information is provided on measures for leveraging investments in managing and reducing the risks. This can be used to identify the most economically profitable risk reduction alternatives.
- The approach enables an overall assessment of risk and highlights the importance of considering the full range of possible outcomes. There are advantages to evaluating the total risk based on the full spectrum of scenarios ranging from low to high probability events. Some advantages derive from the opportunity to understand how different factors influence each component of risk and how they, in turn, affect the total risk. Other advantages relate to the risk-based decision making, as the ranking and prioritization of risk reduction measures may vary depending on whether the measures are evaluated with respect to single or multiple low and/or high probability events. The case study results clearly illustrate the potential sub-optimization that may arise if measures are evaluated only based on individual risk scenarios.

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Analysis of Barriers and Opportunities for Reclaimed Wastewater Use for Agriculture in Europe

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Abstract: This paper presents an analysis of the perception regarding reclaimed wastewater reuse in agriculture conducted in the European Union regions. The analysis is based upon a SWOT framework and applies a cluster analysis to reduce the dimension of the responses enabling an assessment of the different perceptions of water reuse. More than one hundred key actors identified among the regions participated in the evaluation of the relevance of aspects identified. The results indicate some groups of countries according to natural conditions (water scarcity) and the strategic role of agriculture as a key factor to determine agent's perceptions and attitudes. The results indicate that the forthcoming EU regulation of water reuse should focus in the problems of the perceived high cost of reclaimed water for farmers and the sanitary risk perception for irrigated crops by consumers as the critical points for fostering the use of reclaimed water in agriculture and the need for regional implementation of the global regulatory framework.

Keywords: water reuse; reclaimed water; SWOT analysis; cluster analysis

1. Introduction

Arid regions of the world usually have a demand for water that exceeds available resources. The use of reclaimed water is frequently mentioned as a "win–win" solution [1,2]. Previous experience in implementing reclaimed water for agricultural irrigation is satisfactory, especially in water-scarce areas [3], such as Spain, California, Australia [4], Jordan [5], or Italy [6]. Nevertheless, there are still some barriers and obstacles that should be reviewed by [7]. Therefore, water reuse "is considered vital to alleviate the demand on existing but limited water supplies and is gaining impetus throughout the world" [8], also as an alternative water resource to fight droughts and water scarcity [9].

Nevertheless, this opinion should be taken into consideration as wastewater is part of the hydrological cycle and its use in a closed basin where resources are already overallocated (as it is frequent in many regions) may increase exploitation of resources [10]. Additionally, the financial cost or the greenhouse gas emissions should also be considered. The main governance instrument in the EU is the Water Framework Directive (2000/60/EC) (WFD) [11]; the WFD has been successful slowing down the deterioration of water status and reducing (mainly point source) chemical pollution, regarding urban wastewater, 88% of EU wastewaters are subject to secondary treatment although water reuse is still low in the EU [12,13].

The EU included reclaimed water as part of the circular economy. As it is considered in the literature [14], the resources efficiency strategy and several regulations are developed with the aim to foster the use of reclaimed water. Water quantity and quality, including reclaimed water, is regulated by the EU mainly through the following: Water Framework Directive (2000/60/EC) [11], the Urban Waste Treatment Directive (91/271/ECC), the Scheme for Fertilizers (EC2003/2003) [15], or the Nitrates Directive (91/676/EEC) [16]. Closely related to EU water regulation is the Common Agricultural Policy provisions 2014–2020 [17] and the Marine Directive [18]. Additionally, the EU also influences water reuse by strategic documents such as Commission communication on Water Scarcity and Droughts [19], Blueprint for Safeguarding European Waters [20], and the Circular Economy strategy [21]. Finally, several international initiatives like the Sustainable Development Goals included in the UN 2030 Agenda for Sustainable Development include fostering the use of reclaimed water within its goals.

However, the keystone in the implementation of reclaimed water for irrigation is the development of the "Regulation EU-2020/741 Minimum Requirements for Water Reuse" (European Commission, 2018) [21]. This regulation has been recently approved by the EU Parliament and seeks the homogenization of reclaimed water quality standards and water risk management systems for all the EU countries. There is a general agreement about water reuse brings benefits [22,23], but the proposed regulation should be adapted to varying conditions in each of the EU regions [2]. Consequently, a specific strategy should be used to foster reclaimed water in each region. This paper tries to answer this issue, analyzing the perception of the opportunities and barriers that several European regions face in the implementation of reclaimed water for agricultural irrigation.

This paper contributes to identifying regions with similar barriers and opportunities to implement reclaimed water in agriculture. We suggest that it is necessary for the implementation of specific strategies adapted to each regions' characteristics if a satisfactory reclaimed water implementation in agricultural irrigation is sought.

The paper continues as follows, firstly with the material and methods employed in the development of the research; secondly, with the cluster analysis; thirdly, with the results discussion; and finally, with the conclusions.

2. Materials and Methods

This research is based on the empirical work made during the European Project H2020 SUWANU-Europe [24], which proposes an exploratory analysis of the opportunities and barriers facing the use of reclaimed water in agriculture. To achieve it, this paper proposes a Cluster Analysis to know the similarities among the regions participating in the project: Belgium, Bulgaria, France, Germany, Greece, Italy, Portugal, and Spain. SUWANU-Europe departed from the results of the previous EU project (SUWANU) [19], which were used to support our analysis. The research design includes the survey of the relevant stakeholders (farmers, private sector, drinking water suppliers, wastewater suppliers, national and local administration, research institutions, and Non-Governmental Organizations (NGOs)) in the eight countries. In Appendix B is attached the table with the resume of key actors provided in the deliverable 2.1 of SUWANU [25].

2.1. Study Area

Regions included in the survey belong to eight European countries carefully selected to promote the adoption of water reuse strategies. The eight regions were selected following criteria of high technological development, Braunschweig; high water consumption in agriculture, Thessaloniki; high contribution of agriculture to regional economy, Andalusia and Plovdiv; total employment, Thessaloniki and Plovdiv; existing legislation, Andalusia; water stress, Thessaloniki, Tuscany, Antwerp, Limburg, and Andalusia; and high levels of rural population, Occitan, Santarem, Plovdiv, Thessaloniki, and Andalusia [2,4,24,26]. These regions belong to Belgium, Bulgaria, France, Germany, Greece, Italy, Portugal, and Spain. Table 1 illustrates the regional differences regarding urban wastewater treatment plants (WWTP) and related variables.

Regions under analysis differ in size and population. For that reason, we use data available in Table 1 such as the number of WWTP or the total discharge of wastewater allowed, to characterize reclaimed wastewater potential availability. Reclaimed water potential availability in these countries supports the idea of considering it as an alternative water resource, i.e., in some water abundant regions, such as Belgium the volume of treated water exceeds agriculture water demand.

Table 1. Data insight for regions.

Country	BEL	BUL	FRA	GER	GRE	ITA	POR	ESP
Region	Antwerp and Limburg	Plovdiv	Occitanie	Braunschweig	Thessaloniki	Po-River	Alentejo	Andalusia
Number Urban WWTP	108	1	3124	2	12	3579	103	668
% Wastewater treated	84%	76%	99%	100%	n.a.	82%	n.a.	87%
Total discharge (hm^3)	325.0	49.06	353.51	35.50	117.71	n.a.	36.09	698.17
Reclaimed water use (hm^3/year)	0.10	1.08	0.10	20.0	2.27	n.a.	30.88	41.42
Irrigation demand (hm^3/year)	15.79	186.0	1015.0	n.a.	1017.0	4750.0	512.58	4241.12
% Wastewater treated	84%	76%	99%	100%	n.a.	82%	n.a.	87%
% Abstraction/ Resources (*)	19%	5%	12%	12%	7%	24%	6%	26%

Source: SUWANU Europe Deliverable 1.1. resume table [27] and (*) Total water abstraction/Renewables resources. Data from EUROSTAT [28].

2.2. Material and Research Design

The material consists in the responses to a large survey conducted from May to July 2019, in the eight EU member states' regions. Aspects analyzed in the survey are categorized following the SWOT framework dimensions (Strength, Weakness, Opportunity, and Threat).

The proposed structure makes a more flexible comparison of aspects identified among the different regions for two main reasons. Firstly, not all regions have the same concern and expectations about the use of reclaimed water for irrigation. Consequently, aspects identified in each region can vary, making the comparison difficult. This classification respects those singularities and allows the aspects characterization following proposed categories. Secondly, whether all regions follow the same classification, the evaluation of the different aspects will show which categories received more attention in each region making results comparable.

Key actors were identified by the regional group from among members of all sectors related to the topic of reclaimed water in agriculture (policymakers, farmers' representatives, water technology companies, wastewater treatment suppliers, government institutions, and research institutions). Each one of the partners identified its regional key actors.

The identification of aspects involved in fostering reclaimed water for irrigation consisted of a three-step process. The first phase consisted in determining whether aspects identified in the previous EU project [29] were still relevant and proposing new aspects not included that could be relevant nowadays. Secondly, a design phase is conducted using different methods such as workshops, key actor interviews, and brief surveys to key actors. The aim of this phase is the final identification of all the aspects influencing reclaimed water implementation. Finally, the third step consisted in arranging the different aspects pointed in previous phases within SWOT framework dimensions (Strengths, Weakness, Opportunities, or Threats) and the categories explained in Figure 1. This process included a discussion about some results that considered an aspect as a strength or an opportunity at the same time, varying in relation to each key actor's opinion.

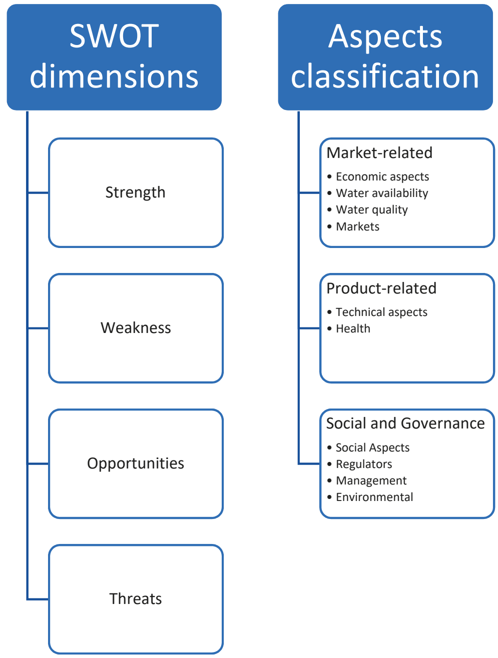

Figure 1. SWOT analysis dimensions and aspects classification proposed.

2.3. Aspects Evaluation

Although the use of SWOT analysis originated in business analysis, it also received uses outside this domain [9], and use of SWOT analysis to identify factors influencing the implementation of reclaimed water has already been made [8]. This paper focused on the evaluation to know the most relevant aspects influencing reclaimed water fostering for agricultural irrigation. The aim of this evaluation is the identification of the most relevant aspects in each region and the comparison of the results among the different regions. The classification proposed in Figure 1 will allow us to compare which groups of aspects have more relevance.

To evaluate aspects relevance, the methodology proposed is a Likert scale from 1 to 5. The Likert scale allows us to evaluate the agreement or disagreement for a series of statements [30,31] and is recommended the use of 5 levels (1 not relevant to 5 very important). This scale allows a neutral option, rate 3, for respondents without a clear answer about a question [31]. Most countries follow a 1–5 scale, although France and Germany use a scale 1–10 that was later converted to a 1–5 scale with the aim to compare results.

The methodology to evaluate aspects relevance also varies from one to another country. The most common tool used was an online survey sent to key users by email in Bulgaria, Greece, Italy, Portugal, and Spain. However, Belgium, France, and Germany evaluated the relevance of the different aspects surveying key actors directly during a workshop. Aspects identified and a preliminary analysis of the main results are available in SUWANU Europe Deliverable 2.1. [25]. In this research, we analyzed the compared results from the different regions following categories explained above (See Figure 1) trying to know which specific characteristics affect the implementation of reclaimed water as an alternative water resource.

3. Results

Generally, SWOT analysis makes a statistical description of the responses with an "expert opinion" for interpretation of the results. Our proposal is innovative as we will use cluster analysis to get some insight into the survey since we have eight countries with different objective characteristics (water scarcity, agricultural demand, etc.) and socioeconomic conditions.

Table 2 shows the results of the survey following the categories classification and SWOT dimensions explained in Figure 1. The higher the value, the more relevant is the aspect. For example, in Belgium, the most relevant categories are product-related strengths; in Bulgaria, strengths related with market-related issues; in France, market-related weaknesses and opportunities; in Germany, market-related opportunities; in Greece, issues about market-related strengths are the most relevant; in Italy, market-related strengths; in Portugal, market-related weaknesses; and in Spain, market-related strengths. This information will be analyzed more in detail following the cluster analysis results.

Table 2. Country average value for each for category for SWOT critera.

Aspects Classification Following SWOT Dimensions	Belgium	Bulgaria	France	Germany	Greece	Italy	Portugal	Spain
Strength Market-related	4.04	**4.20**	3.00	3.56	**4.30**	**4.74**	**4.45**	**4.45**
Strength Product-related	**4.50**	**4.00**	2.83	**4.11**	**3.85**	4.10	4.31	**4.30**
Strength Social and Governance	2.70	0.00	**3.75**	3.59	3.11	**4.54**	4.18	**4.38**
Weakness Market-related	4.23	3.67	**3.83**	3.08	3.06	4.20	**4.94**	3.48
Weakness Product-related	**4.60**	**4.00**	2.33	2.50	3.38	0.00	**4.50**	0.00
Weakness Social and Governance	3.73	3.25	2.25	3.50	3.25	**4.56**	3.86	3.20
Opportunity Market-related	4.05	3.50	**3.83**	**4.50**	2.79	3.92	3.61	3.82
Opportunity Product-related	4.00	3.00	2.50	3.72	3.42	4.21	3.42	0.00
Opportunity Social and Governance	4.06	2.92	2.38	3.42	3.21	3.69	3.63	4.01
Threat Market-related	3.80	3.25	3.50	**3.75**	3.28	3.35	3.70	3.63
Threat Product-related	**4.30**	3.00	1.50	3.08	**4.00**	0.00	4.42	0.00
Threat Social and Governance	3.95	**4.11**	3.38	3.33	**4.51**	4.17	3.71	4.03

Source: Own elaboration with data from SUWANU Europe SWOT Analysis. (1 means: no relevant; 5 means: very relevant).

This preliminary analysis shows that the perception of reclaimed water differs considerably according to each region's characteristics. We want to process this information and try to find similarities and differences that explain the perception of SWOT dimensions among the different regions to know the barriers and opportunities that reclaimed water is facing within each region. Consequently, this research drives a cluster analysis to evaluate which regions face similar barriers or opportunities in implementing reclaimed water for agricultural irrigation. For that reason, we simplify the results (see Table 3) to identify the type of barriers or opportunities the regions are facing. We calculated the average values of the aspects following the classification explained in Figure 1.

Table 3. Categories average evaluation.

Aspects Classification	Belgium	Bulgaria	France	Germany	Greece	Italy	Portugal	Spain
Market-related	4.03	3.66	3.54	3.72	3.36	4.05	4.18	3.85
Product-related	4.35	3.50	2.29	3.35	3.66	2.08	4.16	1.08
Social and Governance	3.61	2.57	2.94	3.46	3.52	4.24	3.85	3.91

Source: Own elaboration.

The analysis of agents' response is difficult to carry out based exclusively on descriptive statistics; therefore, we try some multivariate techniques whose primary purpose is to group objects based on the characteristics they possess. We select cluster analysis because it tries to identify internal homogeneity within the aspects of a group (cluster) and an external heterogeneity between each cluster [32].

We also analyze the differences among the regions following SWOT characteristics; on the one hand we pay attention to the prevalence of positive or negative aspects among the countries (see Table 4).

Table 4. Difference positive minus negative aspects SWOT analysis.

Aspects Classification	Belgium	Bulgaria	France	Germany	Greece	Italy	Portugal	Spain
Market-related	0.06	0.78	−0.50	1.23	0.75	1.11	−0.58	1.16
Product-related	−0.40	0.00	1.50	2.25	−0.11	8.31	−1.19	4.30
Social and Governance	−0.92	−4.44	0.50	0.18	−1.44	−0.50	0.24	1.16

In this analysis, we can observe the prevalence of positive or negative aspects among the regions under analysis. On the one hand, Germany and Spain's key actors give more importance to positive issues in the three categories. On the other hand, Bulgaria, France, and Italy give a more positive relevance to two over three categories, and finally, Belgium, Greece, and Portugal give a higher negative relevance to two over three categories. This analysis could suggest that fostering reclaimed water could be "easier" in Spain or Germany than in Portugal or Bulgaria.

On the other hand, we provide an analysis of the prevalence of internal or external aspects among the countries. SWOT analysis evaluates internal aspects (strengths and weaknesses) and external aspects (opportunities and threats); consequently, we try to show which aspects are more relevant in each region. This analysis' results are provided in Table 5:

Table 5. Internal–external SWOT analysis.

SWOT Aspects	Belgium	Bulgaria	France	Germany	Greece	Italy	Portugal	Spain
Int–Ext	−0.36	−0.66	0.90	−1.46	−0.26	2.80	3.75	4.32

This analysis suggests that internal aspects are more relevant than external ones in France, Italy, Portugal, and Spain, while external aspects are more relevant in Belgium, Bulgaria, Germany, and Greece; we will discuss these results in the following part of the paper together with cluster analysis results.

Finally, cluster analysis is an exploratory data mining technique applied to the whole survey trying to force objects (responses in our case, regardless of the country of origin) to fall into the same group (called a cluster) following a similar definition of distance [32]. Our degrees of freedom "a priori" are eight countries by 12 groups: 4 SWOT dimensions × 3 categories (see Figure 1). We apply principal components analysis to reduce the dimensionality of the space of answers, although the results show that the Kaiser Meyer-Olkin (KMO) is lower to 0.6, recommending the use of original data [32]. Consequently, according to Hair [32], a logical basis is needed to determine the variables to apply cluster. For that reason, this research concludes the proper variables are "market-related, product-related, and social and governance".

According to the results of cluster analysis (see Figure 2), we may identify two cluster groups: (a) Belgium, Portugal, Germany, Greece, and Bulgaria and (b) Italy, Spain, and France. The next section makes a deeper analysis of the perception in these four groups and tries to analyze results.

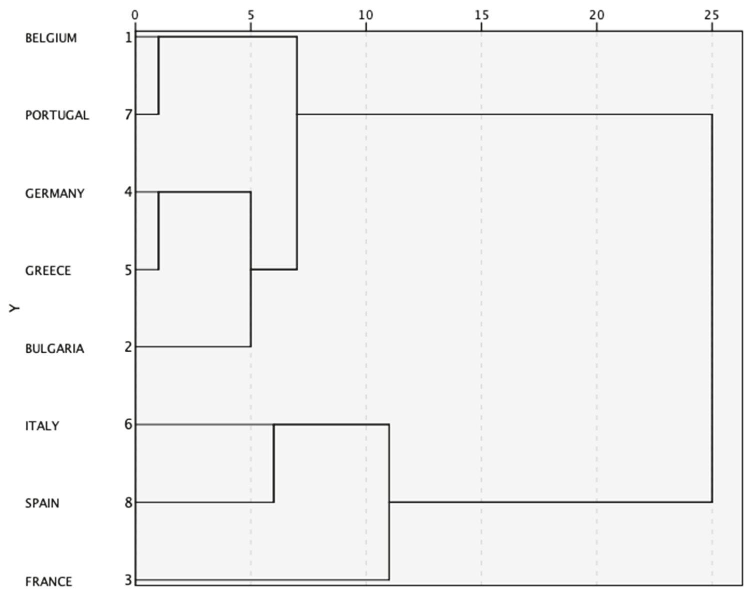

Figure 2. Cluster analysis result.

4. Discussion

This paper seeks similarities and differences among the barriers and opportunities perceived by key actors of eight EU regions. We conducted a SWOT analysis with the key actors' groups established for each one of the regions participating in the project. The first step was the identification of the relevant aspects. The SWOT analysis and the evaluation of the aspects were supported by a cluster analysis to identify similarities and differences among the regions. Following the categories proposed above (market-related, product-related, and social and governance), cluster analysis results in two groups: (a) Belgium, Bulgaria, Germany, Greece, and Portugal and (b) France, Italy, and Spain. An in-depth analysis of the aspects identified within the countries of each group is conducted.

Providing an in-depth analysis of the first cluster group (BE, BU, GE, GR, PT), we focus on the relevance of each category observed in Appendix A, Table A1 (see a resume in Table 6). We also provide a heatmap where the most relevant issues are colored red and the less green in Appendix A, Table A3. This first cluster key actors seem to agree about the high relevance of product-related issues. This can be a reflection of the potential use of reclaimed water supported in the existence of technological and technical conditions to treat wastewater (especially in Germany). However, in the same way, there exist some regions where product-related is also considered a weakness (Portugal and Bulgaria), or weakness and a threat (Belgium). Paying attention to the specific aspects identified by the key actors of those regions, we can identify risks for implementing reclaimed water, e.g., energy cost, the lack of infrastructure to distribute reclaimed water from the WWTP to the crops, or the necessity to learn from most advanced countries (see Cyprus and Israel). It can be observed a kind of consensus about the cost of implementing water reuse and the cost of reclaimed water itself being aspects that should be faced by the public administration within these countries.

Table 6. Belgium, Bulgaria, Germany, Greece, and Portugal categories average evaluation.

Aspects Classification	Portugal	Belgium	Bulgaria	Germany	Greece
Market-related	4.18	4.03	3.66	3.72	3.36
Product-related	4.16	4.35	3.50	3.35	3.66
Social and Governance	3.85	3.61	2.57	3.46	3.52

Trying to understand how to face the cost management issues identified in Portugal, Belgium, or Bulgaria, we can observe that Germany shows just the opposite. German key actors may give higher relevance to product-related issues and market-related opportunities with comments such as "The potential self-financing business model of AV-BS (region of Braunschweig, Germany) where water fees paid by customers to support the system", or, the most relevant opportunity, "irrigation free of pollutants". These aspects are similar to the market-related issues identified in Belgium and Bulgaria, where the cost of reclaimed water for agricultural irrigation is considered a weakness. Moreover, these countries only identified one aspect as product-related strengths, e.g., "knowledge and technology about reclaimed water treatment". Consequently, as we explained just above, Belgium and Bulgaria give more relevance to product-related weaknesses than strengths. Nevertheless, the rest of the regions in this cluster, Germany, Greece, and Portugal, agreed in considering product-related aspects as a strength. These regions considered the existence of previous success stories and technology available, an issue that will facilitate the implementation of reclaimed water. However, it also seems relevant that product-related issues are considered as a threat in Portugal, Belgium, and Greece. In the case of Belgium, this is clear (see above), but in the case of Portugal and Greece, although these countries' key actors considered the existence of technology and technical conditions good to support reclaimed water implementation, they also suggested that the potential nanoparticles could require intensive treatment that threatens the use of reclaimed water. Besides, in the case of Portugal, the lack of infrastructure was not only considered a weakness but also a threat to overcome in the future.

It can be concluded that product-related issues are the most relevant in this cluster, positively such as in Germany or negatively like in the rest of the regions. The position regarding costs is the main difference between these regions. Paying attention to German product-related issues, they are considered the most relevant concerning strengths and opportunities dimensions. The technical experience of AV-SB (the German regional water company) in water reuse and the 4th wastewater treatment technology developed can be considered the solution for the high cost of reclaimed water that is perceived in the other countries. They have previous experience in reusing 20 hm^3 out 30 hm^3 wastewater discharge, and consequently, their cost is lower, but a relevant reason to understand this difference can be that Braunschweig is a small region, with only two WWTPs in comparison with the other, bigger regions with more WWTPs.

It can also be concluded that technology and technical issues to foster the use of reclaimed water for agriculture exist, and key actors within this cluster agreed about it. Nevertheless, energy costs or distribution costs should be overcome. Other aspects also received attention in this cluster. It can be observed how social and governance is considered a relevant threat in Belgium, Bulgaria, and Greece. On the one hand, Belgium and Bulgaria highlight that the new regulation will imply a high cost in implementing reclaimed water. On the other hand, Greece's key actors are more concerned about the public perception itself, e.g., "disagreement between various parties" or "uncertainty in the public . . . ". Portugal considered social and governance issues more a strength than a threat, e.g., their key actors highlight the existence of information programs and a perception of safety in using reclaimed water for agriculture. Finally, as explained above, Germany's key actors did not consider social and governance a relevant category, indeed one of the most relevant aspects identified is the no existence of water scarcity in the region.

Tables 4 and 5 illustrate the point that product-related issues are evaluated negatively in all the regions except for Germany, at the time that market-related issues are evaluated positively among the regions with the exception of Portugal (the most relevant category is market-related weakness, due to distribution costs). Being classified as market-related or product-related, this group is characterized by being concerned about the cost of implementing, distributing, and storing reclaimed water. In the case of Germany, the country is characterized by being able to drive this issue for the last years.

Regarding the second cluster, regions (FRA, ITA, ESP) give relevance to social and governance and market-related strengths (see Table 7). They perceive that the most relevant aspects are related to social and governance issues. This situation shows that society is concerned with water scarcity problems and considered that reclaimed water could help to fight it. Nevertheless, it is important to inform society properly, because threats about public perceptions also received higher attention, even when the new European Regulation implementation, the existence of reclamation standards, and good communication with users are considered a relevant strength to face the use of reclaimed water.

Table 7. France, Italy, and Spain categories average evaluation.

Aspects Classification	France	Italy	Spain
Market-related	3.54	4.05	3.85
Product-related	2.29	2.08	1.08
Social and Governance	2.94	4.24	3.91

In Appendix A, Table A2, it can be observed the evaluation of the different aspects' categories. France, Italy, and Spain give more relevance to internal than external aspects and they all agree to evaluate positively product-related issues (see Tables 4 and 5). It seems that key actors are optimistic about the implementation of reclaimed water in these regions. Paying attention to aspects identified as social and governance strength, the most common relevant category among this cluster, it can be observed that key actors considered the existence of an EU regulation such a quality guarantee to achieve public support. This characteristic opposes to the other cluster, where the EU regulation quality requirements were considered as an "extra cost". Besides, there exists an agreement about water scarcity and the necessity to seek alternative water sources. Consequently, the need for constant water flow for irrigation, the higher water demand for agricultural uses, and the existence of WWTP can lead to the consideration of reclaimed water as a proper alternative water resource. The difference between Greece and these countries can be motivated in the smaller number of WWTP and the greater availability of water regarding irrigated areas (see Table 1).

Finally, other aspects also received a higher score by key actors. For example, both Italy and Spain considered market-related issues as a strength. Aspects identified are related to the existence of quality standard, of constant water flow, or the environmentally friendly consideration of reclaimed water. All these aspects are related to the social and governance issues commented in the previous paragraph. In the case of France, market-related issues are considered an opportunity. The existence of big cities in the coastal areas and the increasing population support this evaluation. This aspect is also the most relevant in Germany, an issue that is supported by previous literature [2]. Finally, social and governance is also evaluated as a threat in France and Spain and as a weakness in Italy. In the case of France and Spain, the lack of a proper communication policy can result in consumers and wholesalers refusing to consume products irrigated with reclaimed water. The same happens in Italy, but in this case, the lack of public support is considered a weakness.

It can be concluded that this cluster is more optimistic than the first one. Although costs are also considered, more attention is paid to social and governance aspects. The motivation can result from a water scarcity situation and the higher water demand for agricultural irrigation. However, the need to communicate properly the benefits of irrigating with reclaimed water is also relevant for the environment and human health. For that reason, the new EU regulation is considered an opportunity within these regions because it is considered a quality guarantee to avoid the distrust from consumers and food chain actors.

The groups that cluster analysis have shown can be seen as counterintuitive as they include only three southern countries (ES, FR, IT) meanwhile Greece and Portugal belong to the other cluster. The relative abundance of water in Portugal and the smaller amount of WWTP in Greece may be an explanation. Besides water abstraction (all uses) divided by available renewable resources in Portugal is closer to Northern countries than to neighboring Spain. Additionally, Italy and Spain have a competitive, export-oriented food industry, which may explain also the differentiation from other countries. Consequently, the relative water scarcity and the competitiveness of agribusiness may explain these results, although further research is required.

5. Conclusions

This paper provides an analysis that identifies the main opportunities and barriers faced by reclaimed water based upon cluster methodology and the interpretation of results. Although regions' hierarchy of topics varies, the global perception is that (a) high cost of reclaimed water for farmers and (b) social fear of products irrigated with reclaimed water should be the keystone of the EU strategy to foster the use of reclaimed water in agriculture.

In our research, we have detected that the perception of key actors varies according to the degree of water scarcity and the importance of irrigated agriculture. France, Italy, and Spain focus on water costs and the necessity to achieve consumer acceptance. Other countries without serious scarcity concerns focus on social governance issues to foster collaboration between farmers and the food chain. Policymakers should consider the impact of new EU regulation and support farmers in the financing of operation, at least in the initial stages, in order to strengthen the risk assurance system that will make transparency and social trust possible. Stronger involvement of regional or basin authorities will be probably the more efficient mechanism to promote water reuse avoiding farers and consumer resistance.

The analysis contributes to identifying the main barriers and opportunities that reclaimed water faces in its implementation process among the different regions. Consequently, when the European Commission seeks the approval of reclaimed water specific legislation, these differences should be considered. As this research concludes, not all the regions considered reclaimed water as an alternative water resource at the same level. In some cases, this is because the cost of water distribution is higher or maybe because there is not enough to achieve public support. Consequently, our opinion, based upon this evidence, is that there is a need for implementing different strategies in the different regions if a satisfactory reclaimed water implementation in agricultural irrigation is sought.

Further research could include other regions within the EU to obtain a complete landscape of reclaimed water barriers and opportunities.

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Appendix A. Tables SWOT Analysis Evaluation

Table A1. Belgium, Bulgaria, Germany, Greece, and Portugal aspects evaluation.

Aspects Classification Following SWOT Dimensions	Belgium	Bulgaria	Germany	Greece	Portugal
Strength Market-related	4.04	4.201 [1]	3.56	4.302 [2]	4.453 [3]
Strength Product-related	4.502	4.003 [2]	4.112 [3]	3.85	4.31
Strength Social and Governance	2.70	0.00	3.59	3.11	4.18
Weakness Market-related	4.23	3.67	3.08	3.06	4.941 [1]
Weakness Product-related	4.601 [1]	4.003 [2]	2.50	3.38	4.502 [2]
Weakness Social and Governance	3.73	3.25	3.50	3.25	3.86
Opportunity Market-related	4.05	3.50	4.501 [2]	2.79	3.61
Opportunity Product-related	4.00	3.00	3.72	3.42	3.42
Opportunity Social and Governance	4.06	2.92	3.42	3.21	3.63
Threat Market-related	3.80	3.25	3.753 [1]	3.28	3.70
Threat Product-related	4.303 [3]	3.00	3.08	4.003 [3]	4.42
Threat Social and Governance	3.95	4.112 [3]	3.33	4.511 [1]	3.71

[1,2,3] represent the aspects with a higher relevance, according to key actors' evaluation per region.

Table A2. France, Italy, and Spain aspects evaluation.

Aspects Classification Following SWOT Dimensions	France	Italy	Spain
Strength Market-related	3.00	4.741 [1]	4.451 [1]
Strength Product-related	2.83	4.10	4.303 [3]
Strength Social and Governance	3.752 [2]	4.543 [3]	4.382 [2]
Weakness Market-related	3.831 [1]	4.20	3.48
Weakness Product-related	2.33	0.00	0.00
Weakness Social and Governance	2.25	4.562 [2]	3.20
Opportunity Market-related	3.831 [1]	3.92	3.82
Opportunity Product-related	2.50	4.21	0.00
Opportunity Social and Governance	2.38	3.69	4.01
Threat Market-related	3.503 [3]	3.35	3.63
Threat Product-related	1.50	0.00	0.00
Threat Social and Governance	3.38	4.17	4.03

[1,2,3] represent the aspects with a higher relevance, according to key actors' evaluation per region.

Table A3. Aspects Relevance Heatmap by Country.

Aspects Classification Following SWOT Dimensions	Belgium	Bulgaria	France *	Germany	Greece	Italy *	Portugal	Spain *
Strength Market-related	4.04	**4.20**	3.00	3.56	**4.30**	**4.74**	**4.45**	**4.45**
Strength Product-related	**4.50**	**4.00**	2.83	**4.11**	3.85	4.10	4.31	**4.30**
Strength Social and Governance	2.70	0.00	**3.75**	3.59	3.11	**4.54**	4.18	**4.38**
Weakness Market-related	4.23	3.67	**3.83**	3.08	3.06	4.20	**4.94**	3.48
Weakness Product-related	**4.60**	**4.00**	2.33	2.50	3.38	0.00	**4.50**	0.00
Weakness Social and Governance	3.73	3.25	2.25	3.50	3.25	**4.56**	3.86	3.20
Opportunity Market-related	4.05	3.50	**3.83**	**4.50**	2.79	3.92	3.61	3.82
Opportunity Product-related	4.00	3.00	2.50	3.72	3.42	4.21	3.42	0.00
Opportunity Social and Governance	4.06	2.92	2.38	3.42	3.21	3.69	3.63	4.01
Threat Market-related	3.80	3.25	3.50	**3.75**	3.28	3.35	3.70	3.63
Threat Product-related	**4.30**	3.00	1.50	3.08	**4.00**	0.00	4.42	0.00
Threat Social and Governance	3.95	**4.11**	3.38	3.33	**4.51**	4.17	3.71	4.03

* Countries belonging to cluster two. Colors represent the less relevant aspects (green) and the most relevant aspects (red), following the average relevance achieved in the survey.

Appendix B

Table A4. Resume of key actors participating in the SWOT analysis, original from D2.1 SUWANU-Europe.

Key Actors' Sector	Belgium	Bulgaria	France	Germany	Greece	Italy	Portugal	Spain
Farmers	2	4		3	3	8		2
Private Sector	1	2		1	4	3		1
Drinking water supplier	1	1						
Wastewater supplier	1	2	1	3		1	2	5
National administration	1	2						2
Local administration	2	4	1	2	3			
Research institution	1	2	1	1	10	3	4	8
NGOs		1				1		6
Total	9	18	3	10	20	15	6	24

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Insuring Water Supply in Irrigated Agriculture: A Proposal for Hydrological Drought Index-Based Insurance in Spain

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Abstract: In Mediterranean-climate regions, irrigated agriculture is especially vulnerable to the risk of hydrological drought and irrigators are particularly concerned about its negative effects. During a hydrological drought episode, irrigators receive insufficient water to meet their crops' water needs, giving rise to the so-called 'water supply gap'. In such circumstances, agricultural production and irrigators' incomes are considerably reduced. In order to minimize the negative effects associated with water supply gaps, a new index-based drought insurance scheme for irrigation is proposed, linked to the variable 'stock of water available in reservoirs'. The proposal, although tailored to Spain, could be easily adapted to other countries or regions because the features of hydrological drought risk are similar worldwide. It is expected that the proposed scheme will improve drought risk management in irrigated agriculture, stabilizing irrigators' incomes and guaranteeing the sustainability of irrigated agriculture in the face of global change.

Keywords: drought risk; water supply risk; irrigation insurance; water use; Spain

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1. Introduction

Agriculture is an economic activity exposed to multiple risks that can negatively influence farmers' income and wealth [1–3]. Particularly notable among all these agricultural risks are production and market risks: production risks such as hail, frost or drought reduce the quantity and/or quality of agricultural production, while market risks are related to potential increases in input prices and decreases in agricultural product prices.

In Mediterranean-climate agricultural regions, such as in Spain, production risks associated with climatological factors pose a more serious threat than market risks [3,4]. This is because this type of climate is characterized by a high frequency of extreme weather events (e.g., hail or heat waves) and by an irregular rainfall pattern, severely affecting agricultural production. Rainfall variability leads to sharp fluctuations in crop yields under rainfed conditions, where precipitations are the only source of water. This is not necessarily the case with irrigated agriculture, where the availability of irrigation water can solve problems related to meteorological drought (below-average rainfall). In fact, irrigation techniques have been developed worldwide as a strategy for both increasing agricultural production and reducing drought-related production risks. It is only when meteorological droughts last for a long time and become hydrological drought episodes (instream flows and reservoir levels below normal) that irrigated agriculture is affected: the availability of water for irrigation is reduced, and thus irrigators cannot fully meet all their crop water needs. This situation, the so-called 'water supply gap', entails notable losses of production and income for irrigation farmers.

In recent years, the changing climate has generated growing concern among irrigators in Mediterranean regions about irrigation water reliability, and they are becoming increasingly aware of the consequences of water supply gaps. Negative predictions about rainfall (lower water availability) and temperatures (higher crop water needs) in these regions could even threaten the sustainability of irrigated agriculture [5]. Furthermore, other global changes, such as increasing pressure from economic activities (mainly industry and tourism), population growth, rising living standards and worsening pollution, are likely to exacerbate the vulnerability of many Mediterranean regions to water scarcity and drought [6,7]. In fact, several studies have found a widespread interest among irrigators in reducing the uncertainty associated with the high variability of their water allotments for irrigation. That interest is manifested in their willingness to pay to reduce the said uncertainty [8–10]. Therefore, there is an urgent need to design new risk-management instruments that could be implemented by irrigators to minimize the foreseen negative impacts of hydrological droughts on irrigated agricultural production [11,12]. Of all the risk management tools suggested to date, insurance has been highlighted as a particularly efficient economic instrument to cope with this risk [13], as it can protect irrigators against financial losses resulting from droughts. In this sense, insurance is viewed as a key instrument forming part of the policy-mix to be implemented in order to adapt irrigated agriculture to climate change; it improves the resilience of the irrigation sector, which is facing increasing uncertainty and vulnerability due to changing climate conditions and other relevant drivers [14]. However, although agricultural insurance is highly developed in first-world countries such as Spain [4,15], the risk of water supply failure in irrigated agriculture is not covered because multiple factors hinder the development of hydrological drought insurance schemes.

Within this context, the main objective of this paper is to contribute to the debate on how to support irrigators in managing hydrological drought risk, by proposing a new insurance scheme for irrigated agriculture capable of overcoming the problems that currently make this risk uninsurable. This scheme is tailored to Spain, as it provides a particularly interesting case study. For this purpose, we exhaustively analyze the factors that currently limit the development of this type of insurance and review previous studies that have proposed insurance schemes to cope with hydrological drought risk. This has allowed us to propose a technically feasible and commercially viable index-based insurance scheme that relies on a variable measuring the stock of water in reservoirs. This scheme is best suited to highly inertial water supply systems, i.e., those with a large water storage capacity compared to annual inflows and annual water demands, as is the case with many basins in Mediterranean and semi-arid developed countries. In order to illustrate how the proposed drought index-based insurance could work in a real-world setting, this paper provides a quantitative example in the Guadalquivir River Basin (southern Spain).

Finally, it is also worth commenting that although the proposed insurance scheme is tailored to Spanish irrigators, the appeal of the developed proposal extends beyond this national scope. Indeed, most of the factors hindering the implementation of hydrological drought insurance for irrigated agriculture are universal, and the technical features of the suggested index-based scheme are applicable in many other countries.

2. Hydrological Drought Insurance as an Instrument for Adaptation to Global Change

2.1. The Risk of Water Supply Gaps in Irrigated Agriculture

As commented above, when faced with a 'water supply gap', irrigators cannot fully meet all their crop water needs because there is not enough irrigation water available in reservoirs and instream flows (hydrological drought episode). The consequences of these water supply gaps in irrigated agriculture can be categorized in three areas: economic (micro and macro), social and environmental. First, from an economic point of view, water supply gaps lessen farmers' income, since they have to choose their crop mixes depending on the availability of water, reducing the irrigated area and/or growing less water-intensive and less profitable crops. On a macroeconomic level, a water supply gap diminishes total agricultural production and value-added of the farming sector and the other related economic sectors, such as the food industry or the agricultural input sectors. In addition, since farmers tend to be risk averse [16], in situations of high uncertainty about the availability of water for irrigation, they tend to reduce the use of inputs [17] and make fewer investments [18]. Therefore, in these situations, economic decision-making by farmers is not efficient from a public welfare perspective.

Second, from a social point of view, water supply gaps in irrigated agriculture may cause a considerable decline in agricultural employment, considering that one hectare of irrigated land generates, on average, 3.8 times more employment than the equivalent rainfed area [19]. In some cases, recurrent water supply gaps may even lead to the abandonment of certain crops that are especially vulnerable to the risk of insufficient water supply, such as fruit groves, which are much more labor-intensive than annual crops.

Third, it is worth mentioning the environmental consequences of water supply gaps: when water allotments for irrigation are below normal, many farmers might illegally extract water resources (i.e., groundwater extractions beyond those legally allowed) to cover their unsatisfied water needs partially. This may aggravate the problems related to the overexploitation of aquifers and surface water bodies (rivers, lakes, etc.) [20,21].

Due to all the aforementioned negative consequences of water supply gaps in irrigated agriculture, it is obvious that uncertainty about the availability of water for irrigation constitutes a major production risk for irrigators in Mediterranean-climate agricultural regions (microeconomic impact). It also poses a problem to society as a whole, taking into account its macroeconomic, social and environmental impacts on social welfare [22]. Therefore, the implementation of proactive measures to adapt to global change in irrigated agriculture must be a priority, in order to effectively manage this climate risk and mitigate its wide-ranging negative impacts [23,24].

Like any other entrepreneur running a business, irrigators are responsible for managing the set of risks that they face. However, public administrations should support farmers' decision-making by establishing an integrated public-private framework (with public incentives) to promote the appropriate adoption of risk management instruments [25]. Adequate management of risks in irrigated agriculture helps stabilize farmers' annual income and also, as previously mentioned, has positive economic, social and environmental consequences for society as a whole.

The strategy traditionally implemented to minimize hydrological drought risk has been to build new water infrastructures such as reservoirs, in order to capture and store a greater amount of water, a strategy known as 'supply-side policy'. However, increasing demands for water and the economic and environmental difficulties involved in enlarging storage reservoirs have ultimately forced policy-makers to stop implementing this kind of supply-side measure. This situation is especially evident in many river basins in Mediterranean and semi-arid regions that are considered hydrologically 'closed' [26,27], where any new demand for water can only be met if the water rights of other users are reduced.

River basin closure is motivating the scientific community and policy-makers to explore new demand-side instruments to manage hydrological drought risk [28]. Among these measures, it is worth highlighting the modernization of irrigation systems, as well as the implementation of water markets, water banks, water option contracts and hydrological drought insurance [29–33]. Moreover, several drought indicators, such as the Standardized Precipitation Index (SPI) [34], the Standardized Precipitation Evapotranspiration Index (SPEI) [35], the Surface Water Supply Index (SWSI) [36] and the Joint Deficit Index for Droughts (JDI) [37], have been proposed; the aim is to calculate these indicators regularly in order to provide accurate information to stakeholders and decision-makers involved in drought management. Of all these demand-side instruments, the hydrological drought insurance has been the focus of very little analysis in the literature as a potential tool for managing hydrological drought risk in irrigated agriculture. Moreover, there are virtually no reports of this kind of insurance scheme having been implemented in a real-life setting.

Despite this scarce analysis and lack of real-world experience, during the last decade the use of agricultural insurance has been promoted by international institutions such as the World Bank, the Organization for Economic Cooperation and Development (OECD) and the European Commission (EC) as a powerful instrument for managing the risks facing farmers, allowing them to transfer the risk of agricultural production to an insurance company [12,38,39]. In this regard, these institutions suggest that it is necessary to insure some risks that are not currently covered, such as the risk of water supply gaps in irrigated agriculture. Furthermore, it is acknowledged that public incentives should be provided in order to encourage the widespread adoption of agricultural insurance as a useful tool for risk management in the farming sector. Public incentives may make this instrument more financially attractive for farmers and for insurance companies. As mentioned above, this public support is justified because adequate risk management in agriculture contributes to enhancing social welfare, through the improvement of agricultural production and the reduction of negative social and environmental impacts.

In this sense, the primary objective of hydrological drought insurance for irrigation should be to guarantee a stable level of income to farmers when their water allotments are lower than their needs. This type of insurance would thus act as a buffer against the microeconomic effects of a water supply gap, reducing uncertainty and allowing farmers to adopt more efficient economic decisions. Moreover, it is worth noting that by achieving this primary objective, this kind of insurance scheme would have a positive effect on rural development since the macroeconomic and social impacts of hydrological drought would also be minimized.

2.2. Factors Hindering the Development of Hydrological Drought Insurance for Irrigators

There are many factors that can explain why hydrological drought insurance for irrigators has not been implemented in a real-life setting [33,40,41]. Some of these factors relate to information asymmetries between the insurer and the insured, which also affect other types of agricultural insurance schemes:

- The 'moral hazard' problem, which arises when the farmer, having taken out an insurance policy, may intentionally behave carelessly regarding the covered risks, with the insurer being unaware of that behavior [42]. This would be the case, for example, with an irrigator who has taken out a hydrological drought insurance policy and, in situations of water supply gaps, does not use his/her entire annual allotment to irrigate. Thus, the irrigator would save costs (lower irrigation costs) and, at the same time, would stand to receive a greater indemnity payment (higher losses claimed).
- The 'adverse selection' problem, that is, those farmers that are most likely to suffer losses are more willing to take out insurance. Since insurance premiums are usually set according to an average measure of risk, low-risk farmers will not be as motivated to insure their activities as high-risk farmers. In this way, a 'self-selection' process occurs where the insurance subscribers would be increasingly likely to suffer losses [43]. This problem creates an actuarial imbalance for the insurer (claim payments greater than premium charges), which inevitably leads to a progressive rise in insurance costs and, ultimately, to the inefficiency of the instrument [44,45].

Both problems could be overcome in a drought insurance scheme for irrigated agriculture by applying the appropriate measures. To this end, it would be necessary to: (a) segment potential insurers (irrigators) according to their level of exposure to the risk of water supply gaps (e.g., dividing them into sub-basin levels) in order to apply premiums corresponding to the risk actually borne by each farmer; (b) implement bonus-malus incentive systems that individually correct the premiums based on farmers' historical behavior; (c) establish deductibles; and (d) apply evaluation rules that audit farmers' behavior in the event of a loss. In any case, it is worth noting that all these measures generate additional transaction costs that affect the cost of the premiums.

Moreover, the design and implementation of hydrological drought insurance for irrigated agriculture present some specific problems that also need to be solved:

- Hydrological drought is a systemic risk [46], which implies that water supply gaps affect a large number of irrigators (all those located in the same river basin) at the same time. Thus, the indemnity payments may jeopardize the solvency of insurance companies. This issue could be addressed, on the one hand, through reinsurance and, on the other hand, by encouraging insurance companies to build up substantial capital reserves. Both measures would also increase the cost of the premiums.
- As already mentioned, climate models predict a decrease in the average volume of water available for irrigation and an increase in its variance, changes that would raise the cost of the premium in a hydrological drought insurance scheme. However, there is a high degree of uncertainty about future changes in the probability distribution function (PDF) characterizing the stochastic variable 'water allotments', which is key information for the insurance actuarial analysis [47]. This problem may be solved by adding an additional ambiguity load into the insurance premium [46].
- The existence of different sources of water supply for irrigation also generates a problem because hydrological droughts do not affect all of them equally. This would be the case, for example, with an irrigator who has rights to surface water, but can also access groundwater sources, reclaimed water and/or desalinated water. In a hydrological drought situation involving cuts in surface water allotments, the irrigator could offset the water supply gap by using any other water sources, such as desalinated water, which is totally secure (this resource is not exhaustible). Under this circumstance, the water supply gap would not be insurable, given the difficulty of complying with the indemnity principle: the insured farmer could benefit from the loss since the indemnity received may be higher than the extra cost incurred for using the other sources of water. Thus, it could be inferred that only those irrigators who have water rights for just one source of supply potentially subject to annual allotment constraints (i.e., a surface water right served by a river basin agency) should be able to take out a hydrological drought insurance policy.
- Farmers (potential insured parties) may influence the decision related to the amounts of water to be distributed among irrigators (water allotments) in each irrigation season. Indeed, irrigators are commonly represented in river basin agencies and take part in the decision-making process regarding the setting of annual water allotments, thus influencing the probability of loss occurrence. This issue makes the risk of a water supply gap uninsurable, since losses due to allotment cuts cannot be considered as entirely accidental.
- In many countries, spot water markets and water banks are allowed during drought periods [30,31]. Thus, the amount of water that irrigators actually use for irrigation in their own farms depends on their economic decisions: whether they decide to sell their water allotment or to buy additional water from other farmers. For this reason, hydrological drought insurance cannot protect against production (crop) losses due to a lack of water, because these losses could be aggravated if the farmer decides to sell his/her water allotment. Thus, a drought insurance scheme should insure the value of the farmer's water allotment, and not the value of his/her crop yield (as in most existing crop insurance schemes). If the insurance scheme is designed this way, irrigators could be protected against lost profits caused by failures in water supplies (reductions in water allotments), regardless of their actual use of water.
- In river basins where reservoirs have a large water storage capacity compared to annual inflows and annual water demands (i.e., inertial water supply systems), hydrological droughts only involve reductions in water allotments after extended periods of meteorological drought, normally longer than a year. In these river basins, it is very difficult to set an appropriate time frame for drought insurance. This is the case with the basins located in southern Spain [48], as well as in most of the basins in Mediterranean and semiarid developed countries. In these inertial systems, if the amount of water stored at the beginning of the hydrological year (in October, when reservoirs reach their lowest water levels after the dry summer) is 'normal' (similar to the average

for this month), there is typically zero probability of a water supply gap in that hydrological year. Consequently, water needs for irrigation are certain to be fully met, even if the year is drier than average. Under this circumstance, no one would be willing to take out insurance covering the risk of cuts in water allotments that year. In these basins, cuts in water allotments would occur only after two or three years with rainfall significantly below the average. Thus, the lower the volume of water stored at the beginning of the hydrological year, the greater the probability of allotment cuts during the year and, therefore, the higher the willingness to take out a drought insurance policy. This calls into question the design of a single-year hydrological drought insurance scheme, similar to other agricultural insurance schemes, because the probability of loss occurrence in a given year is not a statistically independent phenomenon; the probability of a water supply gap differs depending on whether the hydrological year starts with stored water levels above or below the historical average. This fact points to the need for multi-year policies or specific contract conditions that encourage the renewal of policies year after year.

While these limiting factors pose a problem in the design and implementation of hydrological drought insurance for irrigation, some authors have begun to analyze how to effectively overcome them. Although the literature on this topic is scarce, several studies have made interesting proposals about how to address these factors. In the next section, we will review the most relevant contributions.

3. Design Alternatives for Insuring Hydrological Drought Risk in Irrigated Agriculture

There are several types of agricultural insurance schemes (see Table 1). A common classification is based on the risks covered [49]: (a) single-risk or single-peril insurance, for example, to cover the risk of production or income losses due to hail or frost; (b) combined or multiple-peril insurance, which protects against income losses due to several risks, such as hail, frost, and floods; (c) yield insurance, which covers any risk affecting crop yields (income), including all climatic and biological (pests) risks, as well as systemic risks not usually covered by single- or multiple-peril insurance (e.g., meteorological drought in rainfed agriculture); and (d) revenue insurance, covering any risk impacting yields (income losses) or prices (drops in output prices or rises in input prices) in order to guarantee a predefined revenue for insured farmers.

Table 1. Types of agricultural insurance schemes [1].

Classification Criteria	Types
Risks covered	*Single risk* or *single-peril insurance* *Combined* or *multiple-peril insurance* Yield insurance Revenue insurance
Evaluation of the damage	On-field loss assessment insurance *Index-based insurance*
Role of public authorities	Private insurance *Public-private insurance*

[1] The options included in the proposed hydrological drought insurance are marked in italics.

Agricultural insurance can also be classified according to how the damages suffered in the farms are assessed: (a) on-field loss assessment insurance (commonly known as 'traditional insurance'); and (b) index-based insurance (see below). In this sense, it is worth highlighting the study by Ruiz et al. [48] who proposed an in-field income loss assessment insurance in Spain to cover the risk of hydrological drought in irrigated agriculture, very similar to the other agricultural insurance schemes available in this country. However, it should be pointed out that the use of this type of loss assessment for cases of systemic risks (such as water supply gaps) has been criticized because it is difficult and expensive to implement when a large number of assessments must be carried out by qualified experts at the same time [46]. Moreover, this scheme proposal fails to address most of the aforementioned issues related to hydrological drought insurance. In fact, the only problem explicitly mentioned is that of water trading, with the assumption that the implementation of this insurance scheme would involve the prohibition of any water transfers by the irrigators insured; this requirement would be difficult to enforce in a real-life setting because of the existence of informal water markets.

In index-based insurance schemes, damage assessment is carried out indirectly through a variable or 'index' strongly correlated with the contingency covered (income or yield losses), without the need for individual loss declarations and in-field assessments. For example, an insurance scheme to cover drought risk in rainfed agriculture could be based on a cumulative rainfall index. Different types of index-based insurance schemes include the following [50]:

- Yield or income index-based insurance, based on a direct measure, such as the average yields or incomes within the same agricultural region.
- Indirect index insurance, based on one or several variables exogenous to the farms. It is possible to differentiate between: i) climatic index insurance, which may consider variables such as rainfall or temperature; ii) agro-climatic index insurance, which takes into account indicators such as the humidity of the soil; iii) satellite imagery index insurance, for example, those relying on vegetation indexes; and iv) index insurance based on other variables, such as the amount of water stored in reservoirs or reservoir inflows.

Although traditional agricultural insurance (in-field assessment) is the most widespread worldwide, index-based insurance offers a series of advantages that considerably reduce the cost of the premiums [38,46,51]:

- As indemnities are calculated according to the value of an objective and non-manipulable index, it is not necessary to perform in-field damage evaluations.
- Because the indexes used are non-manipulable, the farmers do not have any capacity to influence the result of the value of the index (and the indemnities) through their behavior. Thus, the moral hazard problem is negligible.
- There is a greater transparency in the calculation of indemnities in comparison with traditional insurance, so there is no room for arbitrariness, with conflict resolution costs consequently minimized.
- Moreover, as the information for the index (e.g., information on the variable rainfall) is the same for both parties, the insured and the insurer, adverse selection is less of a problem than with traditional insurance.

However, the major drawback of index-based insurance is 'basis risk'; that is, the risk related to possible differences between the indemnity (calculated according to the index) and the actual loss suffered by the farmer. It may happen that when a farmer experiences a loss on his/her farm, the indemnity payment, based on the exogenous index, is either far above or below the actual loss. This situation may occur if there is not a strong positive correlation between the index measured and the loss experienced. Accordingly, it is evident that the main condition that a variable must meet to be considered as an index is a high correlation with the insured loss, in order to reduce basis risk. Other relevant requirements of index-based insurance are the following [52]:

- The method used to calculate the indexed variable must be available to all potential insured farmers.
- The values to be used in the index must be objective and non-manipulable, and must be regularly made public through appropriate diffusion channels.
- There must be historical records of the exogenous variables used to calculate the index, and information regarding feasible future trends (climate models accounting for uncertainty about future changes in the PDF characterizing the stochastic variable 'water allotments') so that insurance companies can perform actuarial analyses based on the proposed index.

The literature offers few proposals of index-based insurance schemes to cover the risk of hydrological drought in irrigated agriculture. In Australia, Zeuli and Skees [53] designed a rainfall index contract, compatible with transactions in a spot water market. However, the authors did not address the imperfect correlation between the rainfall variable used as the index and the risk of hydrological drought (actual farm losses).

For Mexican irrigators, Leiva and Skees [54] proposed a hydrological drought index-based insurance considering the variable 'inflow accumulation', also demonstrating that index-based insurance is fully compatible with water markets. Similarly, in Spain, Maestro et al. [55] have proposed an index-based insurance scheme to cover the risk of water supply failure in irrigation districts, which is based on reservoir inflows. However, this proposal is suitable only for water supply systems with low inertia (i.e., where reservoirs can only store enough water to meet users' needs for one year or less), where situations of hydrological drought cannot be forecasted at the beginning of the hydrological year. Nevertheless, this is not the case for most of the river basins in Mediterranean and semiarid developed countries with a high risk of supply gaps, where storage capacity is usually large enough to face long (interannual) drought periods.

The most recent proposal is that developed by Maestro et al. [56] for California, which suggests the use of an index currently calculated by the state's water management authority, aimed at estimating the availability of water in the basin to meet users' demand. The value of this index is calculated in early May every year (at the beginning of the irrigation season), accounting for the value of the same index in the previous year and the forecasted runoff for the current hydrological year. As the value of the index is affected by its own value in the previous year, there is a high risk of intertemporal adverse selection. To minimize this problem, the authors propose three alternative insurance designs: (a) 'early bird' insurance (the product must be bought one year before the irrigation season begins); (b) variable premium insurance; and (c) variable deductible insurance.

Finally, it should be noted that agricultural insurance can also be classified depending on the role of public authorities. If the function of the public sector is purely regulatory and in defense of competition, the agricultural insurance system is private. However, mixed public-private systems are also common. Under these public-private partnerships, agricultural insurance is part of the agricultural policy and the state creates the regulatory framework to promote this risk management instrument through subsidies and public reinsurance. The state is motivated to do so in view of the social benefits derived from its implementation [3,38,39].

4. Proposal for a Hydrological Drought Index-Based Insurance for Irrigation in Spain

Under Spanish law, all water resources are in the public domain, and consequently, any private water use (e.g., irrigation or industrial uses) is subject to administrative authorizations or legal concessions (water use rights), which are granted by the River Basin Agencies (RBAs) for extended periods (from 25 to 75 years). However, it is worth differentiating between the volumes set in these concessions or water rights, which theoretically are fixed in order to meet all users' needs, and water effectively delivered to rights holders (e.g., irrigators) each year (water allotments), which depends on the available water stored in reservoirs in the current hydrological year. Thus, the water volume specified in the water rights is only actually available for irrigators in average or wet hydrological years, when total water availability is higher than the aggregated water rights granted. In cases of resource scarcity (hydrological drought years), water resources are prioritized, and domestic water users are served first. For irrigation and other economic uses, the remaining water resources available are allocated proportionally. Thus, irrigators have to deal with interannual variability in their water allotments; they may sometimes receive a much smaller volume than that established in their concessions, and in some years may even receive no allotment at all, generating 'water supply gap' situations. In this regard, a new insurance scheme to cover this type of risk could play a fundamental role in enhancing irrigators' risk management in order to stabilize agricultural incomes.

The Spanish Agricultural Insurance System (SAIS) is one of the most well-developed and successful systems worldwide [4,15]. The maturity of the SAIS is evidenced by its penetration rates: in 2017, more than 234 thousand agricultural policies were contracted, covering 13.8 million hectares (36.9% of the total crop production area), insuring a capital of 9.93 billion Euros (34.6% of the total agricultural production value) [57]. Much of the success of the SAIS is due to the reduction in commercial insurance premiums thanks to the subsidies granted by the public sector. This public support averages 40% of the cost of the policies [57] and, in some insurance lines, reaches up to 65%, the maximum allowed by the European Union. Configured through a public-private partnership [58], the insurance policies offered cover multiple perils in many crop production, including all climatological risks (e.g., hail or frost, and even drought in rainfed crops). However, as previously mentioned, irrigated agriculture remains unprotected against the risk of irrigation water supply gaps.

In this paper, a hydrological drought index-based insurance scheme for irrigated agriculture is proposed as the most suitable insurance design, given the advantages of index-based insurance over traditional insurance, as explained in the previous section. The benefits derived from the lower management and administration costs and greater transparency of this type of insurance outweigh the shortcomings stemming from the basis risk, especially when the index chosen meets the aforementioned requirements.

Another relevant factor justifying the choice of index-based insurance to cover the risk of hydrological drought is related to the procedure followed in Spanish RBAs to set annual water allotments. In Spain, irrigators are represented in the Commissions on Reservoir Water Releases (*Comisiones de Desembalse*). In April each year (at the beginning of the irrigation season, when reservoirs reach their highest storage levels), the Commissions propose to RBAs the water allotments to be set for the irrigation season. RBAs have traditionally considered these proposals, taking into account the active farmers' lobbying activities. The possibility that the irrigators (potential insured parties) may have influenced the probability of loss occurrence (water allotments lower than water rights) made the risk of water supply gaps uninsurable (supply gaps could be artificially generated).

Nevertheless, since the approval of the Basin Drought Plans (BDPs) in 2007, RBAs' annual water allocation decisions must adhere to the action protocols established for that purpose, at least in theory (in practice, these guidelines have not always been fully implemented). Indeed, BDPs, in accordance with the requirements set by the European Water Framework Directive, have established a technical procedure for the distribution of available water resources based on a system of possible water availability scenarios (normality, pre-alert, alert and emergency), minimizing the previously-existing arbitrariness (allocation decisions swayed by pressure from lobbying groups). However, despite the existence of these action protocols, in cases of water scarcity, there is still a certain degree of arbitrariness in the allocation of water [40]. This means that some doubts remain as to whether water supply gaps in Spanish River Basins are an insurable event. This justifies the design of an index-based insurance scheme relying on a transparent and non-manipulable indicator that is highly correlated with the losses to be insured (actual and accidental supply gaps). In this regard, as stated by Brown and Carriquiry [59], Leiva and Skees [54] and Maestro et al. [56], an appropriate index for hydrological drought insurance for irrigation should be based on the river flow accumulation or, as with the approach adopted in this paper, the stock of water available in reservoirs.

In addition, we propose that this hydrological drought insurance for irrigation should be included in the SAIS, with a level of public support similar to the rest of the agricultural insurance schemes in Spain. The SAIS has effectively addressed the problem of covering systemic risks, such as drought in rainfed crops, using private coinsurance and public reinsurance [58]. Thus, it may be deduced that the risk of water supply gaps in irrigated agriculture could also be covered through a coinsurance pool and a public reinsurance program, guaranteeing the viability of this scheme.

Below, we discuss the elements of the proposed insurance contract, following the guidelines summarized in Table 2.

Regarding the *material elements* of the contract, the following design options are proposed:

- *Insurable interest*: contrary to traditional crop insurance, where crop yields are the insurable interest, the proposed contract considers the full annual water allotment (as established in the water right granted by the RBA) as the interest to be insured. Therefore, it is basically a single-risk insurance policy for the farm, but the insurance policy could also be taken out as a complementary coverage alongside the traditional agricultural insurance schemes, if all crops in the farm are insured. In any case, it is worth remarking that a requirement for insuring this type of risk is that the only resources available to insured farmers are from surface water (allotments set annually by the RBA). If farmers have access to alternative sources of irrigation water (e.g., groundwater—wells—or desalinated resources), they can cope with surface water supply gaps by resorting to other sources, and thus do not need insurance.
- *Insured capital* (*IC*): this is the value of the annual full water allotment, constituting the maximum amount of compensation that the insurer would be obliged to pay to the insured in case of extreme hydrological drought. In this regard, it is proposed that this value should be agreed or estimated, on a farm-by-farm basis, as equivalent to the difference between the annual gross margin of the insured farm with full water allotment and the estimated annual gross margin of the same farm under rainfed conditions. These calculations should use data on average gross margins for irrigated and rainfed crops in each irrigation district. Thus, by considering the actual planned crop mix under full irrigation (different for each farm) and the typical crop mix under rainfed conditions (the same for every farm), the value of the annual full water allotment could be estimated for each insured farm.
- *Index*: this is the variable used to determine the occurrence and intensity of the loss. As mentioned above, the index must meet certain requirements; particularly important features are its close correlation with the insured loss and being non-manipulable. Taking into account these characteristics and the analyzed literature, we suggest using an estimate of the stock of water available in reservoirs (*SW*) of the water system (reservoir network) that supplies irrigation water to the insured farmer. More precisely, it is proposed that this index should be calculated annually on 1 May as the sum of the water stored in the reservoirs at the end of the previous hydrological year (30 September) plus the inflow accumulation in the reservoirs from the beginning of the hydrological year (1 October) to the following 30 April. Thus, this index could be calculated annually at the beginning of May and, depending on its value, it can be determined whether there is a loss (high probability of supply gap—reduced water allotment) and, if so, the intensity thereof, as discussed below.
- *Loss*: the occurrence of loss is verified when the value of the *SW* index is lower than a previously determined threshold *T* of water stored. *T* corresponds to the minimum stock of water in reservoirs that would allow the RBA to approve full water allotments. Thus, in order for a loss to be declared, *SW* must be lower than *T*, since under such circumstances there will probably be restrictions on irrigation allotments and, consequently, a loss incurred by irrigated farms. This loss could be 'partial' (*SW* lower than *T*, but higher than *L*, with the latter parameter defined as the lowest limit of water stock that allows the RBA to approve non-zero irrigation allotments), or 'total' (*SW* lower or equal to *L*, which would mean zero annual water allotments for irrigation). Therefore, the declaration of losses (and also the indemnity assessment) could be made at the beginning of the irrigation season, specifically on 1 May, once the value of the *SW* index has been calculated.

- *Indemnity* (*I*): this is the estimated cash amount equivalent to the value of the damages caused by the loss. As it is an index-based insurance scheme, the calculation of the indemnity would not require in-field damage assessment. It would be calculated automatically, after determining the value of *SW*, as follows:

$$I = \begin{cases} 0 & \text{if} \quad SW \geq T \\ IC \times (1 - DED) \times f(SW) & \text{if} \quad T > SW > L \\ IC \times (1 - DED) & \text{if} \quad SW \leq L \end{cases} \tag{1}$$

  As can be seen, the indemnity (*I*) depends on the deductible (*DED*, the percentage of the insured capital that the farmer is responsible for covering) and on the intensity of the loss, quantified by the function *f(SW)*. As already pointed out, according to the indemnity principle, the insurance should only cover the actual damage suffered; under no circumstances should the insured party receive an additional benefit. In this sense, to ensure compliance with this principle, and also in order to lower the cost of the premium, a deductible equal to 30% of the *IC* is proposed. Regarding the function measuring the intensity of the loss, it could be assumed to be linear: $f(SW) = (T - SW)/(T - L)$. This simplified way of measuring the intensity of the loss entails the assumption that the marginal value of the water allotment remains constant despite the volume of water allotted (i.e., the value of one cubic meter of irrigation water is the same regardless of the scarcity of the resource). In any case, other more accurate functional forms assuming an increasing marginal value of water could be used alternatively in order to measure the intensity of the loss.
- *Premium*: an annual policy with an annual premium is proposed.

Table 2. Main features of the proposed hydrological drought index-based insurance in Spain.

Type of Element	Element	Main Feature [1]
Material	Insurable interest	Annual water allotment
	Insured capital (*IC*)	Difference between the annual gross margin of the insured farm with full water allotment and the estimated annual gross margin under rainfed conditions
	Index	Stock of water available in reservoirs (*SW*)
	Loss	$SW \leq L$ (total) or $T > SW > L$ (partial)
	Indemnity (*I*)	0 in case of $SW \geq T$ $IC \times (1 - DED) \times f(SW)$ in case of $T > SW > L$ $IC \times (1 - DED)$ in case of $SW \leq L$
	Premium	Annual
Formal	Contract term	From 30 September year *n* to 1 May year *n* + 1
	Claim	No need for irrigator to inform insurer about the loss
Personal	Insured	Individual (irrigators) or collective policies (irrigators' associations)
	Insurer	Private-public insurance company pooling all private insurance companies involved in the SAIS
	Reinsurer	Public non-profit reinsurance body

[1] *T*: minimum value of *SW* that allows the River Basin Agency (RBA) to approve full water allotments; *L*: lowest value of *SW* that allows the RBA to approve non-zero irrigation allotments; *DED*: deductible as a percentage of the insured capital.

With regard to the *formal elements* of the insurance contract, the following design options are proposed:

- *Contract term*: we propose that the contract is valid from the formalization of the policy during the pre-established contracting period and the corresponding payment of the annual premium (during the month of September every year), to 1 May of the following year, after the calculation of the *SW* index. However, the policy should be extendable for subsequent annuities. In addition, in order to minimize the existence of intertemporal adverse selection, if a pre-alert, alert or emergency situation occurs during the contracting period (September), only policies taken out in previous years would be eligible for renewal. The inclusion of new insured irrigators or modifications in the insured capital due to changes in the crop patterns would only be possible during the contracting period under normal hydrological conditions.

- *Claim*: as it is an index-based insurance, the irrigators do not have to inform the insurer about the occurrence of a loss. On 1 May, the value of the *SW* index is calculated and, if this value is lower than T, the compensation to be received by each insured irrigator is automatically determined.

Finally, in relation to the *personal elements* of the insurance contract, the following options are proposed:

- *Insured*: this could refer to holders of individual policies (irrigators) or collective policies (irrigators' associations including all farmers operating in the same irrigation district).
- *Insurer*: following the SAIS procedure, it is proposed that Agroseguro, S.A. (the insurance company pooling all private insurance firms engaged in the SAIS) should be the only insurer (only one type of insurance contract with the same terms will be on offer), although the policies may be marketed by the various private insurance companies pooled within the SAIS.
- *Reinsurer*: similar to the rest of the agricultural insurance schemes included in the SAIS, we suggest that the proposed scheme be reinsured by the Insurance Compensation Consortium (*Consorcio de Compensación de Seguros*), a public non-profit reinsurance body.

5. Illustrative Example

The explanation of the proposal provided above is now complemented with a quantitative example illustrating how the drought index-based insurance proposed could work in a real-world setting. This example is aimed at further supporting the idea that this kind of insurance would be easily implementable, thereby enhancing the resilience of irrigated agriculture (the impact of droughts could be minimized).

For this purpose, the Guadalquivir River Basin (GRB), located in southern Spain, is taken as a case study. The stock of water available in reservoirs in the GRB at the end of April (index *SW* proposed above for insurance purposes) has already been modeled by Pérez-Blanco and Gómez [33], who reported that the Weibull distribution is the function that best fits historical data:

$$f(SW;\lambda,k) = \frac{k}{\lambda}\left(\frac{SW}{\lambda}\right)^{k-1} e^{-\frac{SW}{\lambda}^{k}} \tag{2}$$

$$F(SW;\lambda,k) = 1 - e^{-\frac{SW}{\lambda}^{k}} \tag{3}$$

where, $f(SW)$ and $F(SW)$ are, respectively, the probability density function (PDF) and the cumulative distribution function (CDF) for the water stored in reservoirs (SW) expressed as a percentage of the maximum value in the historical data, k is the shape parameter and λ is the scale parameter of the distribution. More specifically, for the sub-basin of *Regulación General*, this PDF was fitted using the values k = 1.484 and λ = 0.347, as illustrated in Figure 1.

According to Basin Drought Plans (BDPs), the water storage capacity in reservoirs within the sub-basin of *Regulación General* is 4718 Mm^3, although the historical maximum volume of water stored on 30 April is 4546 Mm^3 (SW = 100%). Average inflows are 3799 Mm^3/year, although there is a large interannual variability (standard deviation = 2808). From this stored water, a demand of 2034 Mm^3/year must be met, most of it (94%) for irrigation purposes (293,762 irrigated hectares using an average of 1909 Mm^3 annually). Hydrological drought events in this sub-basin are determined on the basis of the volume of water stored in reservoirs at the beginning of irrigation season (1 May), using the following thresholds:

1. *Normality*: volume of water stored greater than 3407 Mm^3 ($SW > 75\%$). Under this scenario, no constraints are placed on water allotments for irrigation.
2. *Pre-alert*: volume of water stored between 3407 Mm^3 and 2273 Mm^3 ($75\% < SW < 50\%$). Under this scenario, irrigation water allotments are reduced by between 5% and 30% compared with the normal scenario.
3. *Alert*: volume of water stored between 2273 Mm^3 and 1150 Mm^3 ($50\% < SW < 25\%$). Under this scenario, irrigation water allotments are cut by between 30% and 70% compared with the normal scenario.
4. *Emergency*: volume of water stored less than 1150 Mm^3 ($SW < 25\%$). Under this scenario, irrigation water allotments are less than 30% of the allotments provided under the normal scenario. Moreover, when *SW* is lower than 10%, zero water allotments have historically been approved in order to guarantee water supply to households and for other urban demands.

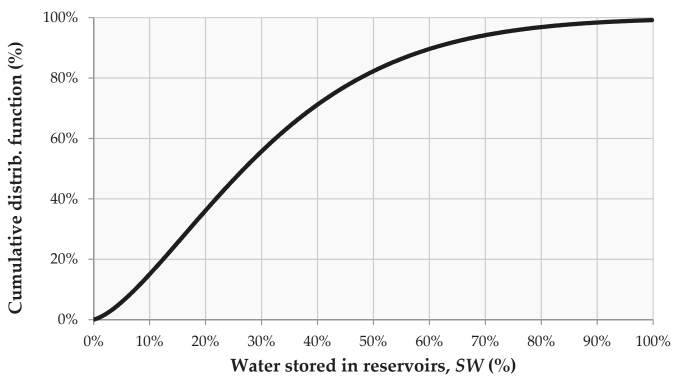

Figure 1. Weibull CDF for water stored in reservoirs in the sub-basin of *Regulación General* (GRB).

This drought management arrangement means that the minimum stock of water in reservoirs on 30 April that would allow the RBA to approve full water allotments for irrigation (*T*) is $SW_T = 75\%$; in cases when $SW < 75\%$ it is expected that these allocations will be cut, which would thus lead to losses in farmers' gross margins. The threshold allowing the RBA to provide non-zero water allotments (*L*) is $SW_L = 5\%$, that is, when $SW < 5\%$ 'total' losses are expected to occur since only rainfed crops could be included in the crop mix.

Under this framework of uncertain irrigation water allotments, consider farmer X, who manages an irrigated farm of 50 hectares located in the Genil-Cabra irrigation district, and whose water resources are those provided by the RBA (surface water resources). When full water allotments are provided, he/she opts for the following crop-mix: olive (40% of farm area), cotton (30%) and corn (30%). However, under hydrological drought conditions (water allotments are cut), he/she must change the crop-mix in order to meet irrigation water needs with the water available. In fact, in extreme cases where water allotments are set at zero, he/she must choose a rainfed crop-mix, maintaining olive under rainfed conditions, and substituting irrigated crops with common rainfed options (wheat and sunflower in the case of the Genil-Cabra irrigation district). Table 3 shows the annual gross margin of farm X with full water allotment and under rainfed conditions (zero water allotment) based on average data from this irrigation district [60]. The difference between the two sets of figures (783 €/hectare, totaling 39,128 Euros for the whole farm area) is an estimate of the value of the annual full water allotment, i.e., the insured capital (*IC*). Similar assessments could be done for any other farm willing to be insured.

Consider that farmer X formalizes the proposed insurance every year during the contracting period (month of September), paying the annual premium set by the insurer. Note that this premium to be paid to the insurer is not fixed in this example since this would require the use of actuarial and financial methods that are beyond the scope of this paper. Thus, the farmer is able to claim an indemnity in the event of losses, calculated according to the value of the index SW on 1 May. In this sense, several scenarios are possible according to expression (1):

- Year n: SW_n = 80%. In this case, since SW_n is larger than T, no losses can be claimed (normal water allocations).
- Year m: SW_n = 50%. In this case, since SW_m is smaller than T, but larger than L, the indemnity to be claimed can be calculated as follows: $I = IC \times (1 - DED) \times \frac{T-SW}{T-L} = 39,128$ Euros $\times$ $(1 - 30\%) \times \frac{75\%-50\%}{75\%-25\%} = 13,695$ Euros.
- Year p: SW_p = 20%. In this case, since SW_p is smaller than L, the indemnity to be claimed equals the insured capital minus the deductible: $I = IC \times (1 - DED)$ $= 39,128$ Euros $\times$ $(1 - 30\%) =$ $27,390$ Euros.

Table 3. Gross margins with full and zero water allotments. Assessment of insured capital (IC) [1].

Full Water Allotment			Zero Water Allotment		
Crop	*Farm area*	*Gross margin*	*Crop*	*Farm area*	*Gross margin*
Olive-irrigated	40%	1003 €/hectare	Olive-rainfed	40%	589 €/hectare
Cotton-irrigated	30%	962 €/hectare	Wheat-rainfed	30%	179 €/hectare
Corn-irrigated	30%	1494 €/hectare	Sunflower-rainfed	30%	219 €/hectare
Whole-farm	100%	1138 €/hectare	Whole-farm	100%	355 €/hectare

[1] Data obtained from Guerrero-Baena et al. [60] for the Genil-Cabra irrigation district.

Of course, it is true that the indemnities calculated following expression (1) may not be exactly the same as the actual losses suffered by the farmers due to allotment cuts (basis risk), but estimated and actual losses are expected to be similar. In any case, as pointed out earlier in this paper, this is the only drawback of index-based insurance schemes, and it is outweighed by the long list of advantages that reduce the cost of the premiums (e.g., no in-field damage assessment required, no room for conflict between farmers and insurers in indemnity assessment, and moral hazard and adverse selection problems are minimized).

6. Conclusions

Due to global change, the failure to guarantee water supply for irrigation is an increasingly pressing risk in Mediterranean-climate regions [61,62]. The severe consequences that hydrological droughts may entail for irrigated agriculture are prompting policy and academic debates about how to effectively manage this type of risk. As supply-side instruments, based on the construction of new water infrastructures, are no longer a viable option in most agricultural regions with mature water economies (i.e., closed basins), the development of new demand-side instruments is needed. In this regard, hydrological drought insurance is a promising instrument for managing the risk of water supply gaps in irrigated agriculture.

In this paper, hydrological drought insurance indexed to the variable 'stock of water available in reservoirs' has been proposed to cover the risk of water supply gaps in irrigated agriculture. This new insurance scheme would be a viable option, since it minimizes many of the problems associated with traditional agricultural insurance, such as adverse selection and moral hazard. It also reduces the cost of the premium because there is no need to make in-field damage evaluations in the event of drought losses. Moreover, the proposed index-based insurance solves certain limitations related to arbitrariness in RBAs' water allotments decision-making. Likewise, the proposed insurance scheme allows this strategy to be combined with other risk management instruments, such as spot water markets and water banks.

The implementation of the proposed insurance in a real-life setting requires additional studies from a supply-side perspective, in order to calculate the commercial premium of the insurance scheme using actuarial and financial methods. Also, further studies from the demand perspective are needed to determine the potential acceptance of this risk management instrument by irrigators through their willingness to pay for such an insurance policy. The comparison of the results of supply- and demand-side studies will allow an exploration of the commercial viability of the scheme proposed. If a significant share of irrigators has a willingness to pay greater than the commercial premium, then the insurance would be viable without the need for public subsidies. In any case, there will be a percentage of irrigators whose willingness to pay is lower than the commercial premium and, therefore, their decision will be to not take out the proposed insurance. Obviously, the existence of public subsidies (and thus cheaper premiums) would mean that a higher percentage of irrigators will choose to take out these insurance policies. In light of this situation, it would be worth analyzing the heterogeneity of irrigators' preferences regarding hydrological drought insurance and the impact of public subsidies on the insurance adoption rate. This information would help support efficient policy decision-making regarding this agricultural insurance scheme.

All the studies outlined above indicate the future lines of research to be developed in order to implement this new insurance scheme and, consequently, improve the economic, social and environmental performance of irrigated agriculture while guaranteeing its resilience in the face of global change.

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