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Sustainability Implications of Deficit Irrigation in a Mature Water Economy: A Case Study in Southern Spain

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Abstract: Deficit irrigation (DI) is an agricultural practice in which the volume of irrigation water applied during the crop cycle is below the irrigation requirements for maximum production, the aim of which is to increase irrigation water productivity. Most research on this technique has focused on agronomic strategies while the economic and environmental consequences have received little attention. This study aims to shed some light on this matter and presents preliminary results regarding the implications of DI with respect to the sustainable use of water resources. The analysis is based on the DPSIR analytical framework (Driving force/Pressure/State/Impact/Response) and the microeconomics of DI. The case study focuses on intensive olive groves in the Guadalquivir river basin in Southern Spain (where olive cultivation accounts for 50% of the total irrigated area). The analysis shows that the widespread use of DI practices, which is the farmers' response to a decreasing net water supply and falling farm incomes (driving force) in the context of a mature water economy, may help to break the DPSIR chain of causality, provided that there are restrictions on any expansion in irrigated area. They can, thus, play a role in achieving sustainable water use. Conversely, demand and supply (regulator) responses involving raising the price of water would lead to higher pressures on the resource and represent a negative driving force in our DPSIR model.

Keywords: sustainability; deficit irrigation; water mature economy; water policy

1. Introduction

To achieve sustainable water management, hydrologic and economic dimensions must be taken into account as part of an integrated management system. This is especially true in parts of the world that have long since entered a 'mature water economy' phase characterised by a limited water supply and growing water demand with increasingly conflicting uses [1]. Some regions have even gone beyond this phase, as indicated by the current 'closed' state of their river basins and/or aquifers. Closure of a water body occurs when all resources have been allocated (and frequently 'overallocated') as a consequence of human activity [2]. Dramatic examples of overallocation can be seen in developed countries such as the USA—specifically California [3]—and Southern Spain [4].

In Spain, irrigated agriculture occupies about 15% of the total cultivated area, but is responsible for 60% of the total value of agricultural production; the average production per hectare of irrigated crops is 6.5 times greater than that of rain-fed agriculture [5]. The specific case of the Guadalquivir river basin (RB) is representative of Mediterranean agriculture and has undergone an intense transformation in the last 50 years, characterised by a significant growth of irrigated areas, a progressive change in the composition of crops and widespread use of deficit irrigation (DI) practices in a context of water scarcity. Against this backdrop, farmers have had to respond to a continued decline in farm incomes, which have decreased by 1.1% annually since the early 1990s, due to the increasing cost of inputs and lower commodity prices [6]. Public responses to water scarcity (once it is no longer possible to increase the supply) in the Guadalquivir RB have focused on improving efficiency, supply control (i.e., irrigation allocation rights and restrictions on irrigated area) and cost-recovery measures. The result has been an increase in water costs (10% real increase for surface water and 15% in the case of groundwater, in the period 1997–2008).

The limited water supply in the Guadalquivir RB, along with the abovementioned increases in the cost of water, have led farmers to use irrigation water more efficiently through the implementation of DI practices. Such techniques allow farmers to maximise value added per hectare and productivity per water input, thus offsetting the rising costs and maintaining farmers' net incomes. Hence, our research focuses on the technique of DI, defined as providing a volume of irrigation water below the maximum irrigation needs (for maximum yield) over the course of the crop cycle and allowing mild stress during growth stages that are less sensitive to moisture deficiency. The expectation is that yield reduction will be limited, and additional benefits can be gained by redirecting the saved water to irrigate other plots or for other beneficial uses [7].

Specifically, this paper focuses on the implications of DI in terms of the sustainable management of the resource, where the use of DI is a response to the decline in farmers' net income (driving force). The general context is that of a mature water economy, and we focus on the specific case of olive cultivation under DI in the Guadalquivir RB in southern Spain (where olive cultivation accounts for 50% of the total irrigated area). In this regard, public initiatives can be understood as a 'response' according to the DPSIR (which stands for Driving force/Pressure/State/Impact/Response) framework [8], while the main driving forces are the need to maintain farmers' net income and the sustanaibility of rural ecosystems. The latter is important given that the expansion of irrigated agriculture has led to increased extractions, which in turn has resulted in a deterioration of the environmental status of water masses (surface-water bodies and aquifers), impacting both the freshwater-dependent ecosystems and human welfare. The public response to this environmental problem has consisted of supply constraints (i.e., allocation rights), restrictions on irrigated area (i.e., a moratorium on new areas), and demand-control policies (i.e., water pricing). On the other hand, in this context of water scarcity and limited supply, the use of DI practices has become the most common private response to farmers' falling income. Using a DPSIR analytical framework, this paper aims to evaluate the effectiveness of these two types of responses (public and private) in terms of achieving sustainable water management.

Unlike most economic models based on the assumption that there is limited availability of irrigated land but an unlimited water supply (i.e., supply significantly exceeds crop needs), this study assumes that water is the limiting factor (a characteristic of a mature water economy). In their model of DI, Berbel and Mateos [9] expanded the model developed by English [10]. We use this expanded model to analyse the farmers' private response, whereby they decide on the appropriate volume of irrigation water for their crops in the analysed context, and the implications of this decision for the sustainable use of the resource. In parallel, we also examine the effectiveness of the regulators' response of increasing water costs (i.e., water pricing) with reference to a DPSIR model, and in the context of the mature water economy and resource scarcity that characterise the case study area.

Our assumption that water is a limiting factor in the RB is consistent with farmers' statements on the matter and is evidenced by the administration's policies in the hydrological plans. Furthermore, in areas where farmers adopt DI practices as the predominant private response, the price elasticity of water demand is also affected, meaning that water pricing becomes ineffective at curtailing water demand unless a disproportionately high threshold price is reached [11]. Thus, our first research hypothesis is that the adoption of DI practices limits the effectiveness of the public (regulator) response aimed at reducing pressures associated with water extractions, thus rebooting the DPSIR chain of causality. Conversely, our second hypothesis to test is that the widespread use of DI practices tends to maximise economic returns to water, thus offsetting the increasing cost of water and preventing a further decline in farmers' net income (the driving force in our DPSIR model).

The structure of the paper is as follows. Section 2 reviews the analytical framework, as well as the main implications of private and public responses to be tested in subsequent sections. The case study and results are presented in Sections 3 and 4, respectively. Finally, Section 5 provides a further discussion of the results, and the main conclusions are summarised in Section 6.

2. Analytical Framework

The process whereby water resource development and use come to exceed available resources has been observed in many parts of the world and has been widely documented in the literature [12–14]. According to Randall [1], a water economy enters a 'mature phase' when the following characteristics occur: (1) inelastic water supply with increasing marginal supply costs (aquifers are already heavily exploited, the best dam locations have been taken and other rivers are protected); (2) high and growing demand (with increasing conflicts among water users); (3) an aging infrastructure which requires expensive renovation; (4) increasing negative externalities; and (5) a rising social cost of subsidising water use. Randall's framework for describing the mature phase of a water economy has been applied to many countries worldwide, including Spain, where a radical Water Law was approved in 1985, reinforcing the public nature of all water resources and ensuring that they are used according to prior allocation of water rights. Based on Randall's model of a mature water economy and in a context of water scarcity, a DPSIR model is constructed to illustrate the sustainability implications of DI. In this regard, a sustainable use of water resources can only be achieved if the resulting responses (public or private) control the driving forces that place pressures on the resource. As discussed in later sections of this paper, this is the only way to break the damaging chain of causality that is the unsustainable exploitation of water resources for irrigation in a mature water economy.

2.1. DPSIR Model

The DPSIR framework, an extension of the PSIR framework, is used to understand the appropriate responses to the impacts of human activities on the environment along a causal chain: driving force-pressure-state-impact-response [15]. Human activities generate pressures on the environment. These pressures change the state of the ecological system, which leads to negative impacts on humans. These negative impacts (should) lead to a societal response. DPSIR analytical models are usually policy-oriented and provide a framework for analysing a problem domain, where all the variables that fall into one of the categories (D-P-S-I-R) have to be included. Details depend on the specific problem domain under consideration, although in the field of integrated environmental assessment this framework has been applied to a wide range of issues, such as coastal zones, water, transport, and pollution control, and more recently, to issues pertaining to sustainable development [16,17].

Thus, in order to gain an overall understanding of the impact of a response measure in a system characterised by interactions between environmental and economic uses, it is appropriate to apply a DPSIR scheme of analysis, as shown in Figure 1. This analytical scheme helps us to understand how the increase in water extractions—a 'pressure' stemming from the 'driving force' of maintaining farmers' income through irrigation development (i.e., area expansion and/or intensification)—leads to changes in the 'state' of water bodies, which become 'impacts', which in turn are met with 'responses'. In this context, the 'pressure' leads to a deterioration in the 'state' of the water body, in terms of both quantity and quality, and this generates an 'impact' on the environment, as well as on human health. The main response to these 'Impacts' has been an increase in water-use efficiency that has driven irrigation modernisation. It should be noted, however, that the concept of 'irrigation modernisation' refers not only to greater efficiency in the use of irrigation water (i.e., through the use of water-saving and conservation technologies (WCTs)), but also to a wide range of socio-economic responses, such as factor productivity maximisation (e.g., widespread use of DI practices, etc.) and public planning and control initiatives (e.g., volumetric billing and water-pricing policies). As expected, these responses generate effects on pressures (e.g., minimising extractions), states (e.g., improving water quality and boosting the quantity available to meet the demands of other users, including the environment) and impacts (e.g., reducing agro-contaminant levels in water bodies). However, such responses may lead to higher costs (e.g., investment and monitoring costs, water price increases, etc.), which have a negative impact on farmers' net income. This may then trigger a new driving force in a potential vicious circle, as shown in our proposed DPSIR framework.

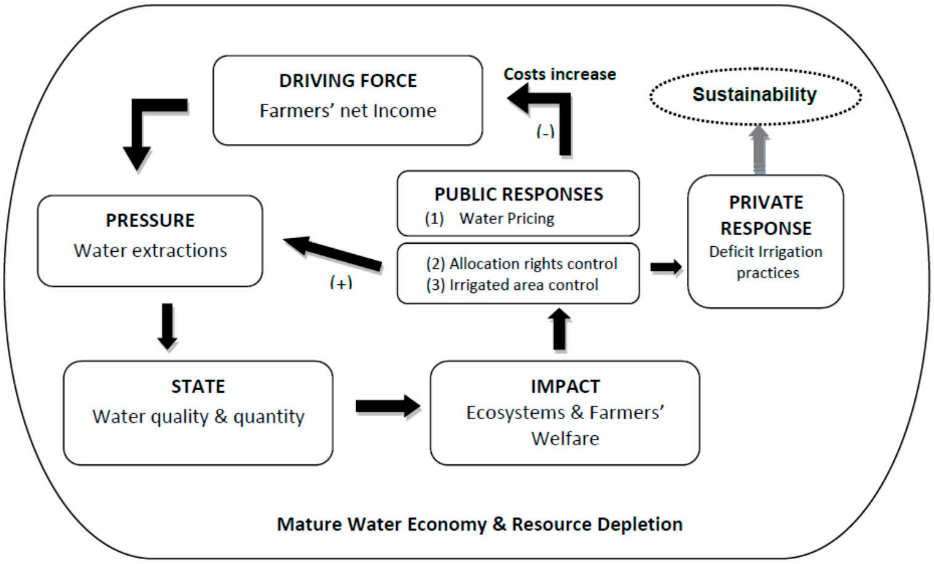

Figure 1. DPSIR model.

Public responses such as control of allocation rights and restrictions on irrigated area may have a positive impact on pressures (i.e., by reducing water extractions). In addition, promoting the widespread use of DI practices as a private response can be a way of achieving more sustainable use of water resources. Nevertheless, public responses based on price increases (i.e., water pricing) imply higher costs for farmers [18], cutting into their net income and accelerating the unsustainable vicious circle depicted in Figure 1. Conversely, private responses (e.g., DI practices) tend to minimise impacts on net income by maximising the economic value per unit of water.

2.2. Implications of the Private Response

Traditionally, farmers calculate the irrigation volume on the basis of the level of evapotranspiration (ET), the effective rainfall, and irrigation efficiency. In turn, irrigation efficiency depends on the uniformity of application and the relative irrigation supply (RIS). RIS is the ratio between the applied irrigation water and irrigation requirements [19] (total irrigation demand less effective rainfall). In the short term, the decision variable that can be managed is the irrigation volume. In their model of DI, Berbel and Mateos [9] expanded the model developed by English [10] to account for DI, by including in the agro-economic model efficiency changes where water rather than land is the limiting factor. Following this development, Expósito and Berbel [20], using estimates obtained from a sample of olive farmers in Southern Spain whose individual responses consisted of the extensive use of DI practices, compared the actual irrigation volume that a farmer applies to a crop with the solutions to the optimisation problem. The findings presented in the next section show that these irrigation practices represent the farmers' primary response and a way to optimise water value (i.e., the productive value per unit of water used), while minimising the effects on farmers' net income of future increases in water-related costs, as well as those stemming from further supply restrictions. As shown in the DPSIR model presented above, if this hypothesis is confirmed, the widespread use of DI practices could potentially break the chain of causality. A sustainable management of the resource could thus be achieved in a context where no expansions in irrigated areas are permitted (i.e., where there is a moratorium on new irrigated areas).

2.3. Implications of the Public Response

Spanish water policy, which is regulated by the Water Act of 1985 and has been adapted to ensure compliance with the EU Water Framework Directive (WFD) of 2000 [21], requires the implementation of a river basin management plan, which must be updated every six years, according to the WFD provisions. So far, two river basin management plan cycles have been defined [22]. The main measures adopted by the government, following a public participation process, include investment in WCTs, volumetric metering, and water-pricing implementation (cost recovery) and, in particular, a moratorium on new irrigated areas. All of these measures can be considered 'public responses' in the mature phase of a water economy.

Once all available water resources have been allocated and a moratorium has been declared on new irrigated areas, water pricing has usually been considered an appropriate public response to guarantee a sustainable use of the resource. Further, its aim is to ensure the price reflects the economic and social value of the resource and contributes to efficient allocation of the resource to different uses [23]. Water pricing has also become the main strategic public tool for water and environmental policy in developed economies, as noted in the WFD [21] and the Blueprint to Safeguard Europe's Water Resources [24]. Most of the related water pricing literature focuses on analysing farmers' responsiveness to pricing pressures. Nevertheless, there is a growing body of literature that concludes that water pricing is not particularly effective at curtailing water demand. The studies of Bernardo and Whittlesey [25], Ogg and Gollehon [26], Dinar and Letey [27], and Varela-Ortega et al. [28] are good examples of attempts to model responsiveness to water pricing among farmers facing restricted water supply. Moreover, several studies claim that irrigation water demand is inelastic below a threshold price, and elastic beyond it [29]. Thus, substantial price increases would be required to produce a reduction in demand, and such increases may have a critical impact on farmers' net income and welfare, as well as on the sustainability of rural environments.

In our case study in Southern Spain, as in many parts of the world, farmers do not freely decide on the amount of water they will use to irrigate their crops, as water access is restricted by water rights (i.e., irrigation quotas). Under conditions of water scarcity and low water prices, the amount allocated is likely to be below the amount of water that farmers would be willing to take at the prevailing price, thus promoting the use of DI practices. This use of irrigation volumes that maximise returns to water [9,20] would, therefore, also push up the theoretical threshold price, after which point demand begins to show negative price elasticity [11]. This point is especially significant in our case study, as public responses that entail increases in water-related costs would only translate into reductions of farmers' income, with little or no impact on reducing water demand. Thus, if this hypothesis is confirmed, and as shown in the proposed DPSIR model, the chain of causality would be rebooted, leading to an unsustainable management of the resource and an endless vicious circle.

3. Materials

3.1. Case Study

The case study is in the Guadalquivir RB, which is representative of the Mediterranean region, and contains 23% of the total irrigated area in Spain with more than 850,000 hectares [5] (see Figure 2). The Guadalquivir is the longest river in Southern Spain (mainly located within the region of Andalusia), with a length of about 650 km and a basin extension of over 57,527 km^2. The basin has a Mediterranean climate with an average rainfall of 630 mm, occasional periods of prolonged drought, and an average annual temperature of 16.8 °C. Annual renewable resources are estimated at 7.1×10^9 m^3 for surface waters and 2.6×10^9 m^3 for groundwater. Olive (both rain-fed and irrigated) is the predominant crop, accounting for more than 50% of irrigated area in the RB. The use of DI practices is widespread. Moreover, the introduction of WCTs has also allowed farmers to significantly increase tree densities (between 250 and 300 trees per hectare), which in turn has enabled them to maximise the economic value generated per unit of water used and maintain (or even increase) their net income.

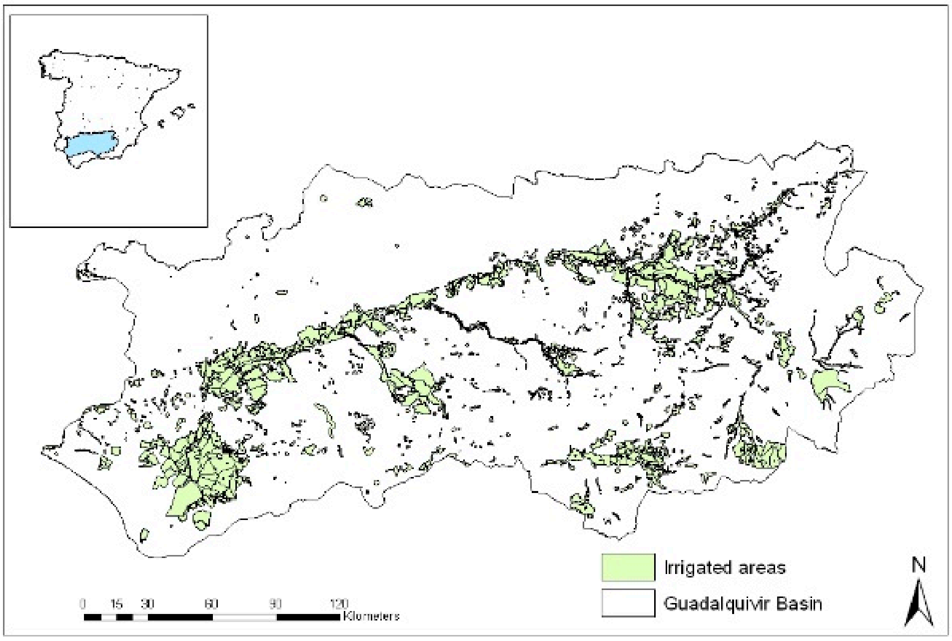

Figure 2. Map of Guadalquivir RB.

3.2. Survey Description

The field work was conducted in the spring of 2014, using information provided by farmers of intensive olive groves in the Guadalquivir RB area regarding yield and irrigation volumes per ha for the period 2010–2013. The analysed sample consists of 48 observations (farmers), which significantly describe this crop in the RB. Descriptive statistics are shown in Table 1, revealing following average values: (a) farm size: 40 ha; (b) density: 283 trees/ha; (c) total allocation of water rights: 2723 m^3/ha (referred to as the legal water quota owned by the farmer); and (d) irrigation doses: 1028 m^3/ha. We observe a discrepancy here, as average water use represents 38% of average water rights (1028/2723), which we consider an indication of the dominant DI strategy studied in our research. Potential ET in the year of the survey was estimated at 492 mm for the intensive olives.

Although the variability within the sample (as shown by the estimated standard deviations) seems high, the table shows that the observed farmers tend to apply a far smaller irrigation volume than that permitted according to their assigned water rights, generally indicating a preference for DI techniques.

Table 1. Basic descriptive parameters.

	Area (ha)	Density (Trees/ha)	Age (years)	Irrigation Rights (m^3/ha)	Yield (kg/ha)	Irrigation Vol. (m^3/ha)
Average	40	283	15	2723	6382	1028
St. Dev.	64	80	6	1846	2344	388

Source: The authors, based on survey results.

4. Results

Our study uses an alternative approach based on the elicitation of a subjective 'perceived production function' as the microeconomic foundation of a farmer's decision-making process regarding water use, rather than the 'objective' production function traditionally used in the literature. Using this individual subjective water demand function elicited from an individual subjective water-yield curve in the 'normal' agronomic range (maximum yield should be within the normal range for the crop and region) as defined in Expósito and Berbel [20], the effectiveness of private and public responses defined in Section 2 will be tested. The answers given by farmers regarding their expectations as to water consumption (m^3/ha) and yield (k/ha) in three possible irrigation scenarios enable the estimation of the quadratic production function needed to test both responses—private (by farmers) and public (water price increases implemented by government authorities)—and their implications for water management sustainability.

4.1. Private Response

This paper aims to test the hypothesis that in the context of a mature water economy (characterised by water scarcity and growing water costs), the farmers' response tends to be to maximise returns to water as the limiting factor in order to maintain their net incomes.

Our study compares actual irrigation volumes applied by farmers with the three optimal solutions to the profit maximising problem as set out by English [10]. Most water-use models are based on the assumption that there is limited availability of irrigated land but water supply is unlimited (i.e., it significantly exceeds crop needs). Accordingly, water is treated as a variable input and land as a constrained resource. This assumption implies that farmers displaying rational economic behaviour should maximise the following profit equation:

$$Z = P_y Y - P_w W - C \tag{1}$$

where Z denotes profit, P_y is the sale price of the crop, P_w is the price of water, and C represents fixed costs. In their model of deficit irrigation, Berbel and Mateos [9] expanded the model developed by English [10] to account for deficit irrigation, efficiency changes and the situation in which land is not a binding constraint and water is a limiting factor. Thus, farmers who behave rationally in an economic sense seek to maximise total net income:

$$Z \cdot A = A \cdot \left(P_y Y - P_w W - C\right) \tag{2}$$

In this equation, farmers may distribute the irrigation dose over a larger irrigated area "A", which is considered a variable in the maximisation process. The fixed factor here is water volume "V", which is constrained so that irrigated area "A" is determined by the total volume of available water "V" (depending on well capacity, water rights, etc.) and the chosen water dose "W".

$$A = \frac{V}{W} \tag{3}$$

Optimal water use is then defined by the value of W that optimises Equation (2), subject to Equation (3).

$$-A \cdot \frac{\partial Z}{\partial W} = Z \cdot \frac{\partial A}{\partial W} \tag{4}$$

Further, quadratic water response production and cost functions, such as those represented below, are estimated:

$$Y(w) = a_1 + b_1 W + c_1 W^2 \tag{5}$$

$$C(w) = a_2 + b_2 W \tag{6}$$

The solution to the optimisation problem posed in Equation (2) considers water as a limited input while land becomes a freely variable input. This alternative model gives the maximum return to water (dose "W_w") which is determined by solving Equation (4):

$$W_w = \left(\frac{P_y a_1 - a_2}{P_y c_1}\right)^{1/2} \tag{7}$$

Table 2 shows data regarding the following variables: estimated irrigation volume that maximises returns to water (W_w), usual DI volume applied (W_u, as declared by farmers), average irrigation effectively applied in the period 2010–2013 (W_o) and irrigation rights allocated to each farmer (2014).

Table 2. Farmers' irrigation behaviour.

m^3/ha	Elicited Values		Survey Values	
	Max. Return to Water (W_w)	Irrigation Rights	Usual DI (W_u)	Avg. Vol. 2010–2013 (W_o)
Maximum	2731	7000	2500	2500
Minimum	248	200	600	600
Median	1013	1750	1450	1042
Average	1163 [1,2]	2723	1357 [1]	1103 [2]
St. Dev.	*571*	1846	*425*	*350*

[1] With a 95% confidence interval, the *t*-test for the difference between means determines that mean W_u = mean W_w;
[2.] With a 95% confidence interval, the *t*-test for the difference between means determines that mean W_o = mean W_w.
Source: adapted from Expósito and Berbel [20].

As Table 2 shows, the average irrigation volume is close to the volume which maximises returns to water when water is the limited resource. A simple *t*-test of significance between the mean values of the data distributions for W_o and W_u, and that obtained from the estimated distribution of variable W_w, shows that data pair W_o and W_w, as well as the pair W_u and W_w, have similar distributions with statistically equal mean values. This result would seem to indicate that our farmers display similar behaviour (on average) to that corresponding to the maximisation of the returns to water, thus confirming our hypothesis about farmers' private response, as described in Section 2.

Therefore, results obtained from the estimated microeconomic model would seem to confirm that olive farmers tend to maximise value generated per unit of water in a scenario where water is the limiting factor, as is the case in a mature water economy. The widespread use of DI practices, which do not deplete allocated irrigation rights (as shown in Table 2), would then function as a desirable private response in our DPSIR model. Using such techniques tends to alleviate the growing pressures in terms of water costs and, thus, minimise the negative impacts on farmers' income, helping to achieve sustainable management of the resource. Nevertheless, this response only leads to a sustainable outcome if the irrigated area remains constant. If the area under DI expands, the pressure on the scarce resource would increase, leading to non-sustainable results.

4.2. Public Response

As previously discussed, the responses to water scarcity included in the Guadalquivir RB management plans have principally focused on the following measures: a) restrictions on irrigated area, b) control and reduction of water rights, and c) water-pricing policies (including volumetric metering and cost recovery). These measures have been incorporated in both RB plans (2009 and 2015), with a special emphasis on controlling the allocation of water rights to irrigators and a progressive implementation of water-pricing initiatives.

As described above, the answers given by farmers regarding their expectations as to water consumption (m^3/ha) and yield (kg/ha) in three possible irrigation scenarios (extreme DI, standard DI and full irrigation), allow us to estimate Equation (5). Based on this subjective production function, farmers' water demand is equal to the marginal product value of water (MPV) defined by the following equation:

$$\mathrm{MPV} = P'_y \times dY/dW = P'_y \times (b_1 + 2 \cdot c_1 W) \tag{8}$$

In the above function, P'_y is equal to the 'net farm gate price' (i.e., the sale price of olives minus harvesting costs) and parameters b_1 and c_1 depend on each farmer and each subjective production function (subject to $b_1 > 0$; $c_1 < 0$). Thus, a subjective water demand function can be elicited for each individual farmer yielding the threshold price at which a farmer's water demand function becomes price elastic (as described by Expósito and Berbel [11]).

Table 3 shows descriptive statistics of the elicited threshold levels (prices) for the estimated marginal product value of water at the standard irrigation volume applied by each farmer in our sample. As observed, elicited threshold prices are well above the current water costs borne by the farmers. Specifically, the average estimated marginal product value associated with the average volume of irrigation water applied in our sample shows that the threshold price would be around 1.2 EUR/m^3. This marginal value of water has been estimated using an average price for olive oil of 2.0 EUR/kg in the period 2010–2013. From 2016, low harvests and high demand have pushed the price of olive oil up to 4.0 EUR/kg and, thus, the marginal value of water has doubled, although the forecast is that prices will level out at around 3.0 EUR/kg in the medium term.

The average marginal value of water (using conservative estimates of product prices) is 10 times the current average water cost paid by our sample of farmers (0.11 EUR/m^3). This would indicate that the price elasticity of water demand is very low (or inelastic), until the threshold level is reached. As a result, the practical effect of public responses is simply cost increases for farmers and a reduction in their net income, thus potentially rebooting the pressure on the resource (as shown by the DPSIR model in Section 2).

Table 3. Elicited threshold price vs. water cost.

	Elicited Threshold Price (EUR/m^3)	**Water Cost (EUR/m^3)**
Maximum	4.68	0.30
Minimum	0.20	0.05
Median	1.03	0.08
Average	1.22	0.11
St. Dev.	0.88	0.09

Note: Average is the mean of all elicited individual parameters. Source: adapted from Expósito and Berbel [11].

Though elicited threshold prices are influenced by the existing water management practices in the RB, a widespread use of DI techniques usually leads to changes in the economic value of water, characterised by an increase in the marginal product value of water and the threshold price level [11]. Consequently, the marginal product value of water and the threshold price determine the structure of water demand and evolve independently of water cost (and related public responses). Therefore, in our irrigated olive case study, public measures resulting in higher water costs would not be expected to play a key role in determining our farmers' water demand, unless water price levels increase disproportionally and rise above the threshold price.

5. Discussion

The farmers' primary response to decreasing net incomes in a context of water scarcity and increasing water costs has been to maximise the productivity of water by using DI as a technique to maximise returns on water. The adoption of such practices has led to changes in private decisions concerning irrigation volumes and the economic exploitation of irrigation water. This response has been developed in the context of a mature water economy, with public constraints on the irrigated area and the allocation of water rights (as specified in the last Guadalquivir RB plans).

Governmental RB plans have also devoted financial resources to promoting 'irrigation modernisation', defined as reducing water losses and increasing water efficiency through automatic control, volumetric metering, etc. Undoubtedly, these public measures have had significant impacts on the use of water resources (i.e., surface and ground waters). Nevertheless, it is beyond the scope of this study to examine those impacts. A more detailed analysis of the consequences of modernisation at the farm level can be found in Fernández, García et al. [30], and at a basin level in Berbel et al. [31].

Regarding the restrictions on irrigated area, a more flexible approach was adopted in the last plan (approved in 2015) in response to lobbying from farmers during the public participation phase. As an exception, the latest RB plan allows the use of an additional 20 hm^3 (mainly from reused treated wastewater) in the case of DI olive groves, which are considered a priority crop under this measure, thus allowing a small increase in irrigated area of around 1.5% (approximately 13,000 new irrigated hectares on top of the existing 880,000). In our opinion, and based on our analytical framework, this relaxation of the moratorium jeopardises the sustainability of water resources in the Guadalquivir RB, as the water in question should be devoted to environmental uses. Conversely, the irrigated-area control should be carried out through more extensive volumetric control of water abstractions, especially with regard to groundwater.

Most experts on the matter argue that the strategic public response, in addition to restrictions on irrigated area and water-pricing measures, should be to store the 'saved water' for environmental purposes and to improve supply reliability instead of expanding irrigation [32]. The public response in the Guadalquivir RB has for the most part (with the exception of the abovementioned reuse of wastewater) followed this strategic line. Regarding price measures, they have been based on cost recovery principles and volumetric pricing, which are commonly understood to be solutions to the increasing pressures on the resource. However, our analysis concludes that they will be ineffectual at reducing water abstraction when DI is widespread (as is the predominant situation in our case study). In fact, only a few marginal areas with very deep aquifers may be affected by an energy cost increase, but the increase in public water prices (presently around 0.02 Euro per m^3) will be mostly useless. Our conclusion is in line with the analysis of the case in Morocco presented by Molle [33], who concludes that the marginal value of irrigation water is far higher than its costs to the farmer and, as is commonly found in many field studies, overly high prices may also push farmers to shift to groundwater, especially where aquifers are shallow. In fact, a recent worldwide review has found no evidence from any country in the world that water pricing has ever been successful at reducing pressures on the resource [34].

Control of water abstraction from surface water is relatively well managed as it generally depends on public storage and distribution systems. On the contrary, there is insufficient control of groundwater abstraction, and this issue should be urgently addressed by the RB authority, as it has been demonstrated that the exclusive use of water pricing (or energy cost) is not enough to produce sensible behaviour in this regard. The main problem in the RB is groundwater abstraction and the solution to this, given the difference between marginal water value and water cost, may well be the implementation of the so-called 'aquifer contracts' [35]. Aquifer contracts seek to ensure the co-management of groundwater between the users and the administration. These contracts are fairly simple in some respects (the water accounting, for example) but more complex in others (stipulating actions that ensure the control of irrigation expansion). Again, international experiences provide only few examples of successful co-management, such as Eastern La Mancha in Spain [36] or Murray in Australia [36].

Sustainable governance of water resources should involve effective responses to control the driving forces and address the pressures on the resource, thus ensuring long-term sustainability. Nevertheless, effective governance remains an important challenge, especially with regard to groundwater resources [37–39]. In our opinion, this challenge can only be successfully tackled with an adequate understanding of the attributes of the resource and its socio-economic context, as well as the explanatory factors for human actions and responses regarding the exploitation of water resources. Only then can an analysis of the impacts of these actions and responses be carried out. As Ostrom [40] points out: "If the initial set of rules established by the users, or by a government, are not congruent with local resource conditions, long-term sustainability may not be achieved. Studies of irrigation systems ... suggest that long-term sustainability depends on rules matching the attributes of the resource system, resource units, and users." We hope this study has helped in that direction, offering a preliminary analysis of the impacts of human decisions (public and private) regarding the use of irrigation water in a context of a mature water economy.

6. Conclusions

The past twenty years have seen substantial progress in the practical application of DI techniques for both annual and perennial crops [41]. Most related studies examine the agronomic technicalities of the optimal DI supply, whereas the economic and environmental consequences have received little attention. This study has presented preliminary results regarding the implications of DI for the sustainable use of irrigation water. To do so, it relies on a DPSIR model and the elicitation of optimal water use and threshold price levels for a sample of olive farmers in Southern Spain.

Findings in our specific case study in the Guadalquivir RB illustrate that the private response to increasing water costs and a fall in farmers' net income in the context of a mature water economy—where the RB authority has implemented measures to further control water use and declared a moratorium on new irrigated areas—has been the widespread of DI practices. Such practices involve reducing the volumes of irrigation water applied in order to maximise the monetary value obtained per unit of water used. As discussed, this response may help break the DPSIR chain of causality, subject to strict control of irrigated areas, and thus help achieve sustainable water use. On the other hand, regulators' responses based on water pricing cannot prevent a rebound effect in this process, which would lead to higher pressures on the resource by negatively affecting farmers' net income (thus reactivating the driving force).

We believe that the extensive adoption of DI practices may have further relevant consequences for RB management, as well as for the sustainable management of water resources, especially in mature water economies (or closed river basins), where a decisive role will be played by farmers' appropriate responses to the 'pressures' and 'impacts' identified. Going beyond the scope of this preliminary study, further research is needed to provide a more comprehensive analysis of the development of these private responses and their impacts on the sustainable use of the resource at a RB or regional scale.

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Abbreviations

The following abbreviations are used in this manuscript:

DI	Deficit Irrigation
DPSIR	Driving force; Pressure; State; Impact; Response
ET	Evapotranspiration
RB	River Basin
RIS	Relative Irrigation Supply
WCTs	Water-Saving and Conservation Technologies

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