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Environmental Concerns in Water Pricing Policy: an application of Data Envelopment Analysis

Giacomo Giannoccaro[1] and Julia Martin-Ortega[2]

Water management is subject to conflicting economic and environmental objectives, and policymakers require a clear overview of the different outcomes derived from different water management options. The aim of this paper is to assess the efficiency of several irrigation water pricing policies with a special focus on their environmental implications. Irrigation is chosen here as a crucial sector of water use in large parts of southern Europe, where pressure over the resource is expected to increase due to climate change. A novel methodological approach to perform an ex ante analysis of alternative water pricing policies is proposed here, where environmental and technical performance are simultaneously considered. This approach takes place in two steps: the first is a simulation of alternative water policies through a mathematical programming model, and the second is the analysis of results by using the Data Envelopment Analysis (DEA) technique. A case study is applied in Puglia (southern Italy), where irrigation is the primary factor of strategic relevance for policymakers regarding water management. Our results show, on the one hand, that alternative pricing policies perform similarly in terms of technical efficiency and environmental efficiency. On the other hand, inefficiency appears to depend mainly on technical rather than environmental concerns. According to the assigned weights, through the DEA technique, the highest improvement for inefficient options may be obtained by better labour use. We conclude that the proposed approach may be a comprehensive and versatile framework for water policy analysis, offering a tool for supporting the decision-making process.

Keywords: Irrigation, Policy assessment, Efficiency, Data Envelopment Analysis, Linear Programming

JEL Classification: Q11, Q15, Q18, Q25

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1. Introduction

Water management is very often undertaken under conflicting objectives. This is particularly the case in areas of water scarcity, where competition for the resource easily brings economic return in conflict with environmental protection. The agricultural sector is the main driver of water use in the Mediterranean regions, with share ranging from 50 to 60% of fresh water bodies (Dworak et al., 2007) and rising to over 80% in certain areas. Water demand for irrigation is increasing, and other sectors are also expected to increase consumption in the near future, when climate change is expected to enhance water scarcity in regions currently under water stress (IPCC, 2007). On the one hand, water management plans for the future need to comply with environmental criteria to assure ecological sustainability. On the other hand, economic viability will always be a necessary requirement for agriculture sustainability and the provision of related social services. Policymakers require a clear overview of the different outcomes deriving from alternative water management policies, and tools need to be improved for supporting the selection of most suitable instruments to specific situations. As a consequence, it is necessary to define a methodological approach and a set of criteria according to which the effects of water policy are to be measured.

At the European level, the Directive EC/60/2000 (WFD) aims at achieving the good ecological status for all water bodies for which it request from the Member States to Program of Measures for River Management Plans (article 11) to bridge the gap between the current status of water bodies and the ecological goals (article 4). The WFD also requires the implementation of water tariffs that imply the recovery of the 'full services' costs (article 9). Selecting the appropriate scientific tools to assess water policy measures and thereby support water management decisions under complex circumstances has been named as one of the major challenges with regard to the implementation of this European norm (Messner, 2006).

The WFD states that economic analysis shall "make judgements about the most cost-effective combination of measures in respect of water use [...] based on estimates of the potential costs of such measures"(Annex III-b). This has lead to the proposal of Cost-Effectiveness Analysis (CEA) as a general method for decision-making in the WFD context (WATECO, 2003). Although the statements and reports of the WATECO group have only a recommendatory character, they have gained acceptance in most national guidelines (for example, see Interwies et al. (2004) for the German case and MIMAM (2008) for

Spain). CEA has become a preferred method for the selection of policy measures in the WFD implementation process.[3]

Two main limitations of the CEA approach have been pointed out in this respect: i) the need for all outcomes to be expressed in monetary units (water management typically involves many non-market factors that are not easy to assess); and ii) the difficulty of achieving a fair distribution of resources among stakeholders (Hajkowicz and Higgins, 2008). In the specific context of the WFD, Messner (2006) argues for the homogeneity assumption regarding measurement effects and their costs and for the existence of multiple water-related benefits and objectives as limitations for the CEA.

Multi-criteria decision analysis (MCDA) where conflicting criteria are accounted for to obtain a compromise have been proposed by a significant portion of the literature as a better alternative in terms of the information that can be derived for policy-making for environmental and resource management (see, for instance, Barreiro-Hurlé and Gómez-Limón, 2008; Jacobs, 1997; Martínez-Alier et al., 1998; Moran et al., 2007; Munda, 1996). Multi-criteria analysis is based on the acknowledgement that economic agents behave bytrying to satisfy several objectives (not just financial profit) that can even be in conflict. The optimal decision satisfies a series of goals associated with those objectives. The MCDA framework ranks or scores the performance of alternative options according to multiple criteria, which are typically measured in different units.[4]

Comparative studies on the application of different applications of MCDA in water resource management have shown that different methods are in close agreement and that there is no clear advantage of any single technique (Gershon and Duckstein, 1983; Ozlekan and Duckstein, 1996; Eder et al., 1997). The main limitations of MCDA relate to the methods of preference elicitation and the selection of criteria and decision options (Hajkowicz and Collins, 2007). However, some of these methods, known as non-parametric methods, such as the Data Envelopment Analysis (DEA), do not require *a priori* assumptions about preferences. In addition DEA evaluation overcomes the *trade-off* or compromise among the conflicting objectives, taking into account the efficiency as criteria for the options ranking.

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[3]The literature is starting to show some progress in this sense. For example, DEFRA in the UK has already published a preliminary analysis of the cost effectiveness of the WFD (DEFRA, 2007), and Berbel et al. (2009) have applied it to a case study in Spain. Van Engelen et al. (2008) reported on two CEA case studies performed in The Netherlands and Demark.

[4] The multi-criteria approach has already been pointed out as a necessary requirement in water planning for many years (see Cohon and Marks, 1975; Hipel (1992) and Lee and Wen (1996)) and has been heavily applied for water policy evaluation. A wide range of multi-criteria methods are used (see Hajkowicz and Collins, 2007 for a review of the research on water planning).

Raju and Kumar, (2006) include the DEA technique into the MCDA methodology, in which the relationship between all inputs and output are taken into account simultaneously. With the DEA, the weights of the assessed measures of policies are those that maximise the ratio between the weighted output and the weighted input.

The aim of this paper is to assess the efficiency of several water pricing policies for irrigation, with a special focus on the environmental implications of the different options. We define an aggregated indicator of efficiency where environmental and technical performance is simultaneously considered. In particular, this research proposes a methodological approach to perform an *ex ante* analysis of alternative water pricing policies, consisting first of a simulation of alternative water policies through a regional multi-agent linear programming model and second on the analysis of these results using DEA. This approach emphasises the principle that, to achieve a certain output, the production process necessarily also produces some undesirable outputs (pollutants or emissions). Therefore, the efficiency is calculated as the ratio between the weighted sum of multiple desirable outputs and undesirable outputs. The paper deals with two aspects of efficiency. The first, the technical efficiency, depends on the optimal allocation of the resource to the most profitable crops (*ceteris paribus*). The second, the ecological efficiency, considers the externalities caused by the irrigated crops on the environment. In both cases, the water pricing scheme will be successful if it will induce an increase of output or the reduction of the externality by consuming the same volume of water. Alternatively, the policy is successful if the same output or externality is produced, with less water.

The broad aim of the research is the development and testing of tools for a better design of water policy management options in the specific context of the European Water Framework Directive.

The research shows an application of the methodology to an irrigated area in the Mediterranean region, specifically in the Foggia region (Puglia) in southern Italy.

The remainder of this paper is as follows. In the next section (section 2), the DEA methodology is discussed in relation to the consideration of eco-efficiency, reviewing the existing literature. In Section 3, the conceptual model proposed in this research is described. The case study is described in Section 4. Section 5 presents the main results, from which conclusions are drawn in Section 6.

2. The Data Envelopment Analysis and eco-efficiency

Data Envelopment Analysis (DEA) is a technique developed to evaluate the efficiency of a number of producers. The typical statistical approach to evaluate the efficiency is characterised as a central-tendency approach, which evaluates producers relative to an average producer. In contrast, DEA is an extreme-point method and compares each producer with only the 'best' producers.[5] This methodology is useful whenever there is no information about the relative importance among outputs or inputs, as it does not require assumptions *a priori* (Callens and Tyteca, 1999). Additionally, DEA deals with variables regardless of their unit measure, provided that these units are the same for every entity (Coelli et al., 1998).

The DEA is frequently used to measure the efficiency of decision units, such as firms, industrial plants, and governmental departments (see, for instance, Glass *et al.*, 2006; Bono and Matranga, 2005; Korhonen and Luptacik, 2004). The DEA technique has also been applied as a useful methodology for ranking irrigation planning alternatives with mutually conflicting objectives (Raju and Kumar, 2006). In their research, the authors apply DEA to select the most suitable irrigation planning alternative in the context of the Sri Ram Sagar Project in Andhra Pradesh (India) using simulated data. However, the authors do not include environmental objectives, which, as mentioned, can be in conflict and in any case need to be accounted for in the new European water policy frame.

The first non-parametric analysis to compare multiple desirable and undesirable outputs is reported in Färe et al. (1989), in which a data set of 30 US paper mills using pulp and three other inputs to produce paper and four pollutants is used. Their results show that the performance rankings of units turned out to be very sensitive to whether undesirable outputs were included. However, the general emphasis on the environmental issue occurred later (Tyteca (1996) presents an exhaustive literature review), and externalities have usually been treated as 'undesirable outputs' of the production process (e.g., Fernandez-Cornejo, 1994; Tyteca, 1996; Piot-Lepetit et al., 1997).

Another possibility is to envision the undesirable outputs as inputs (see Seiford and Zhu, 2002 for either approach). De Koeijer et al. (2002) modelled environmental effects as conventional input. Efficiency is of particular interest when related to specific inputs that cause environmental impacts, such as pesticides and fertilisers. Even more interesting is the extent to which the associated environmental impacts can be reduced by more efficient use.

[5] For a more in-depth discussion of DEA, the interested reader is referred to Cooper et al. (2000) or the seminal work by Charnes et al. (1978).

Korhonen and Luptacik (2004) propose to measure the eco-efficiency of 24 power plants in Europe in two different ways. In the first approach, they measure the eco-efficiency in two steps. First, technical efficiency and the so-called ecological efficiency are estimated separately. Then, the results of both models are taken as the output variables for the new DEA model (with the inputs equal to 1), which provides the indicator for eco-efficiency. Although this approach considers undesirable outputs as an input, environmental concerns are actually run separately. In the second approach, they attempt to build up a ratio that simultaneously takes into account the desirable and undesirable outputs. Three different models are estimated: i) the first model is based on the idea of presenting all outputs as a weighted sum but using negative weights for undesirable outputs; ii) the second model considers the externalities as input, and thus the efficiency results as the ratio of desirable output to a weighted sum of undesirable outputs and inputs; and iii) a model in which the ratio of the weighted sum of the desirable outputs minus that of the inputs to that of the undesirable outputs is considered.

The authors found that both approaches (i.e., separate and simultaneous) achieve almost the same result in terms of finding the most efficient plants, although the ranking for all power plants is slightly different. However, the second approach provides a deeper insight into the causes of the eco-inefficiency and shows the potential improvement with respect to the particular inputs and outputs. Namely, from the weight of the inputs, it is possible to obtain an indication for the importance of particular inputs (undesirable). In a similar manner, the magnitudes of the outputs (desirable) are extended.

In the literature, there is no universally accepted DEA approach (Alder et al., 2002 for a review). However, as Alder et al. (2002) conclude, "… whilst each technique may be useful in a specific area, no one methodology can be prescribed as the panacea of all ills".

Two problems, broadly speaking, are argued concerning the DEA technique: i) the weak discriminating power when the number of options under evaluation is small; as a consequence, too many options are scored as efficient, and ii) the weights are sometimes practically unreasonable or undesirable (Li and Reeves, 1999).

To our knowledge, only one DEA model, in the strict sense defined above, i.e., starting from the ratio of the weighted sum of multiple desirable outputs to undesirable outputs has been applied until now to the measurement of the environmental performance of water pricing (Giannoccaro et al., 2008). However, it should be noted that the authors perform the calculations for technical efficiency and ecological efficiency separately: the comparison of eco-efficiency is made among for performance of the local agricultural system under different water pricing hypotheses.

In this paper, we propose a novel approach in which the efficiency estimation includes all environmental and technical items simultaneously. We consider fertilisers, water and pesticides as environmental inputs; therefore, the ratio of the weighted sum of the economic and environmental outputs to the production and environmental inputs also acts as an indicator of efficiency accounting for environmental concerns. This approach enables the analysis of the nature and causes of all inefficiencies. As pointed out by Korhonen and Luptacik (2004), positive externalities can also be included as desirable environmental outputs. All authors mentioned here have measured eco-efficiency as the ratio between technical desirable outputs and only environmental undesirable outputs. Instead, by our understanding, water irrigation also includes environmental externalities, mainly linked with land conservation, that have a positive impact on the environment (OECD, 2001).

3. Conceptual model

The first step of the proposed model consists of the simulation of alternative water-pricing policies through a mathematical programming model (see appendix). In a second step, we proceed with the analysis of the results by DEA method.

The simulation of the policy is performed by modifying water tariffs. From the simulation of each policy, the most significant variables are selected and categorised as inputs, desirable outputs, and undesirable outputs. These variables that will be analysed by DEA can be classified into two typologies: economic and environmental items. Table 1 show the variables necessary for running the DEA model.

Table 1- Variables for running the DEA analysis

Conventional Resources					
	Input				Output
	Land	Labour	Capital	Water	Added Value
Unit of measurement	10^3 hectares	10^3 hours	10^6 Euro	10^6 m^3	10^6 Euro

Environmental Externalities*			
	Desirable outputs	Undesirable outputs	
	Land cover	Pesticides Risk	Nitrogen surplus
Unit of measurement	10^6 days	10^3 Kg	10^6 t

* For environmental externality indicators see Berbel and Gutierrez (2005)

Let us maintain n water pricing policies. Their effects are simulated through the mathematical programming model at the first step, and the common amount of m inputs and k outputs are estimated. Suppose that m input items and k output items are selected according to Table 1. In particular, for $m=1,2,...i$, the subscript for production inputs is assigned, while for $m= i+1,i+2,...p$, the environmental inputs (undesirable outputs) are specified; at the same time, for $k=1,2,..,r$, the subscript for conventional outputs is identified, while for $k=r+1, r +2,.., q$, the desirable environmental outputs are specified. Therefore, in the case of the implementation for each of these $j= 1,…, n$ water policies, we have the vector m_{ij} of the overall inputs and the vector k_{rj} of the overall outputs.

At a second stage, we apply DEA for efficiency estimation. DEA is a technique based on the application of a linear programming algorithm aimed at finding the most suitable weights for each variable such that the ratio of outputs on inputs of several data sets is made as close as possible to 1. Once the most suitable weights are found from each data pattern (performance of a decision making unit, or policy options), the relative efficiency index is calculated (the most efficient being equal to 1, while the others have a lower index).

Then, for each water pricing policy, we form the virtual input and output by (yet unknown) weights (v_i) and (u_r):

$$\text{Virtual input}= v_{\text{i}}\, m_{\text{i}j}+\ldots+ v_{\text{i+1}}\, m_{\text{i+1}j}+\ldots+v_{\text{p}}\, m_{\text{p}j} \qquad (1)$$

$$\text{Virtual output}= u_{\text{r}}\, k_{\text{r}j}+\ldots+ u_{\text{r+1}}\, k_{\text{r+1}j}+\ldots+u_{\text{q}}\, k_{\text{q}j} \qquad (2)$$

Then, the weights using the DEA (CCR model) technique (Charnes et al., 1978) that maximise the ratio Virtual output/Virtual input subject to

$$\text{v}_i\, \text{m}_{ij}+\ldots+ \text{v}_{i+1}\, \text{m}_{i+1j}+\ldots+\text{v}_k\, \text{m}_{pj} =1 \qquad (3)$$

are determined.

In addition and according to the CCR model formulation, the equation is set such that the ratio should not exceed 1 for every policy, and weights are non-negative.

The weights are chosen in a manner that assigns a best set of weights to each policy. The term ‘best’ is used here to refer to a solution in which the resulting input-to-output ratio for each policy is maximised relative to all other policies (when these weights are assigned to these inputs and outputs for every policy).

DEA efficient policy j^* obtained as an optimal solution among the n policies results in a set of optimal weights (v^*, u^*). The ratio scale θ* is evaluated by

$$\theta^* = \frac{\sum_{r=1}^{q} u_r^* k_{rj}}{\sum_{i=1}^{p} v_i^* m_{ij}} .$$

(4)

As mentioned earlier, (v^*, u^*) are the set of most favourable weights for policy j^* in the sense of maximising the ratio scale (4). v_i^* is the optimal weight for the item I, and its magnitude expresses the relevance, relatively speaking. Furthermore, if we examine each item $v_i^* m_{ij}$ in the virtual input (1), then we can see the relative importance of each input. The same situation holds for u_i^* and the virtual output. These values not only show how each item contributes to the evaluation of policy$_j$ but also to what extent they do so.

4. Case study

4.1 Case study description

To illustrate the proposed approach, a case study located in the Foggia province in southern Italy is carried out. This region is the second most important agricultural area in Italy, after the Po Valley, and is mostly devoted to durum wheat (about 25% of the national production), vineyards, olive orchards, and horticultural crops (including about 30% of the national production of processed tomatoes). Irrigation water is the primary factor of strategic relevance for farmers and a social priority for policymakers.

This area is characterised by a Mediterranean climate with cold wet winters and hot dry summers. Rainfall varies from less than 400 mm/year to more than 700 mm/year, but there are also recurrent periods of drought. As a consequence, water availability depends strictly upon the weather conditions.

The area covers 442,000 ha, of which almost 32% can be potentially irrigated. Management of the irrigation system is under control of the Irrigation Board, named *Consorzio per la Bonifica della Capitanata* (CBC). The CBC water supply campaign goes from April to November, and on average the system conveys about 106,000,000 m^3 of freshwater. The irrigation system consists of a network of underground pipelines through which high-pressure water is conveyed to final distribution points and from which farmers may directly attach their irrigation devices (mostly drip irrigation systems). The water supply is available on demand.

At present, water is allocated through a system of water rights. In most cases, water rights are based on the historical use of the resource by the farmers. The right is non-tradable and thus gives a sort of advantageous position. In the case of water shortages, a relevant quantity of water is diverted from irrigation to industry and urban and domestic uses, with non-compensation given for farmers' losses in revenue.

The CBC applies a volumetric pricing method, including an increasing tiered rate system aimed at a 'fair' allocation of water volume among landowners whose fields are served by the conveyance system.

Aside from the water conveyed by the CBC, the usage of water from other natural sources (most of which is underground water) is also very extended and is estimated to cover about 50% of the overall irrigation water (INEA, 2001). Farmers drill private wells and set all of the necessary equipment for lifting, storage, and delivering water to the crop fields. This source of water is particularly fragile, and it is currently monitored and controlled very little by water authorities. In fact, a reform to control the excessive exploitation of natural resources leading to irreversible environmental degradation began in 2008.[6]

4.2 Water pricing options

It can be expected that direct pricing methods, such as the volumetric rates, are to be preferred due to their efficiency. However, the implementation cost and the running costs of the volumetric pricing method are higher than those related to other methods. At the same time, the WFD requires that all costs for water services should be covered and ecological goals (among others, minimum flow rate[7]) should be accomplished.

Five pricing schemes were designed to deliver (within existing constraints and case study characteristics) the same outcome. First, we looked for the same objective function outcome of the mathematical programming model and water consumption, avoiding any charge increase. The P0 subscript designates this scenario, and the water pricing options are *P0.Vol_tot, P0.Input, P0.Output and*

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[6] This reform was introduced by the Regional Law No 9 on May 2008 (PUGLIA, L.R. n. 9/2008).

[7] The main objective of the WFD is to restore a good ecological status for all water bodies across the European community by 2015. The definition of good status includes targeting the use of renewable resources to reach a minimum ecological flow. Essentially, this is defined as the minimum water flow to preserve the natural or 'non-altered' status of most of the relevant ecosystems (e.g., natural habitats and wildlife).

P0.Area as alternatives to the current scheme, P0.Baseline. Second, we looked for the same outcome in terms of the water saved, namely increasing the charge and reducing the current water consumption at the basin level (about 30%) (P1 subscript). Ten options of water pricing were simulated as a whole. The main features of the pricing schemes are reported as follows:

Baseline: Water pricing currently consists of a fixed annual fee per hectare (around 15 €/ha) and increasing block tariffs in the case of exceeding consumption (0.09 €/m^3 up to 2,050 m^3/ha, 0.18 €/m^3 for an additional 950 m^3/ha, and, from 3,000 m^3/ha, 0.24 €/m^3 without any volume restriction). The water from other sources (non-CBC) is currently free of charge, although farmers have to pay the costs of extracting and pumping the water to their irrigation systems at an estimated average private cost of 0.09 €/m^3, which includes the cost for lifting, storage in a reservoir, and pressurising the irrigation system.

Vol_tot: The current three-tiered rate system for CBC water is maintained. In addition, the introduction of a single rate volumetric method for the non-CBC water is hypothesised. Due to the lack of existing estimations for this rate in the literature, a tariff of 0.03 €/m^3 is assumed.

Input: To reflect an indirect environmental tax on irrigation practices, this regime involves a price surcharge on inputs (e.g., plants or seeds, consumable irrigation equipments, ferti-irrigation materials), regardless of the actual water consumption from either source. The surcharge is calculated on the basis of average water consumption and is different for each crop. This pricing method is intended to encourage the cultivation of crops requiring lower water input.

Output: Water consumption is charged proportionally to the gross return from irrigated crops, regardless of the water source. The charge rate applied to each crop is calculated as the ratio between the current value (according to the CBC tariffs) of its specific water consumption and the corresponding gross return.

Area: A per-hectare charge is set equivalent to the average CBC cost per hectare of irrigated area (82 €/ha), and the water volume available to each farm is fixed; farmers are still expected to maximise the acreage of irrigated crops. It is still a relatively easy method to be implemented and is also easily understood by farmers.

The policy scenarios do not involve a change in institutional assets or the re-allocation of water rights, and the resource saving is only aimed at the accomplishment of the environmental standard. The assumption is that, to be effective, the WFD should divert some of the water to the environment, according to the minimum flow-rate requirement.

The case study was investigated in the context of a lack of change in the technology and the alternative water supply. The transaction costs are assumed to be negligible. We acknowledge this to be a limitation of our study.

5. Results

The first results of the model consist of the most relevant outcomes of the optimal solutions found through mathematical programming. In Table 2, the production inputs and all desirable outputs and undesirable externalities are listed.

Table 2 – Results of water pricing policy simulations

Pricing policy	Objective's function	Production input				Desirable output		Undesirable output	
		Land	Labour	Capital	Water	Added Value	Land cover	Pesticide Risk	Nitrate surplus
Unit measurement	10^6 EUR	10^3 hectares	10^3 hours	10^6 EUR	10^6 m^3	10^6 Euro	10^6 days	10^3 Kg	10^6 t
P0.Baseline	607.42	400.25	25.83	208.94	195.81	657.13	37.61	711.13	29.47
P0.Vol_tot	602.96	400.25	25.83	211.17	195.81	657.13	37.61	711.13	29.47
P0.Input	607.31	393.81	22.01	175.26	195.81	616.92	36.28	649.29	27.40
P0.Output	602.81	393.81	24.03	188.08	195.81	630.91	40.91	759.26	27.18
P0.Area	609.09	412.25	23.26	200.47	195.81	625.87	37.07	785.07	29.41
P1.Baseline	546.42	399.48	21.35	195.44	148.34	588.92	36.35	667.94	28.23
P1.Vol_tot	517.67	393.81	20.27	189.73	118.82	625.87	35.58	615.31	27.51
P1.Input	493.73	379.81	16.35	139.86	148.26	492.65	31.59	592.92	24.51
P1.Output	471.62	385	15.29	162.02	118.82	444.08	33.42	533.44	25.52
P1.Area	504.04	406.77	19.74	212.59	148.26	492.65	35.86	713.3	28.33

Changes in water pricing policies induce farmers to adopt different farm strategies and thus different performances of their agricultural systems may be result.

First, we analyse only technical efficiency. This approach is the simpler way to assess the water policy options. In Table 3, the results of the DEA are shown, and optimal weights are listed.

Table 3- Technical efficiency score and optimal DEA weights

Pricing Policy	Technical efficiency	Weights				
		Land	Labour	Capital	Water	Add Value
P0.Baseline	1.00000	0.00165	0.00000	0.00000	0.00174	0.00152
P0.Vol_tot	1.00000*	0.00165	0.00000	0.00000	0.00174	0.00152
P0.Input	1.00000	0.00022	0.00000	0.00324	0.00176	0.00162
P0.Output	1.00000	0.00102	0.00000	0.00168	0.00145	0.00159
P0.Area	0.97302	0.00125	0.01426	0.00000	0.00078	0.00155
P1.Baseline	0.99541	0.00136	0.01550	0.00000	0.00085	0.00169
P1.Vol_tot	1.00000	0.00000	0.00000	0.00000	0.00842	0.00178
P1.Input	1.00000	0.00000	0.00000	0.00438	0.00261	0.00203
P1.Output	0.99582	0.00102	0.00000	0.00168	0.00145	0.00159
P1.Area	0.86374	0.00068	0.03283	0.00000	0.00051	0.00177

* Weakly efficient according to the slack value

A total of 5 options out of the 10 simulated are equally efficient. Concerning scenario *P0* (no charge increase), all options (expect for the 'area' system) are technically efficient. The *Vol_tot* scheme is efficient, but it shows a slack for the capital value amounts to 2.23 units. Things slightly change if we analyse scenario *P1*, in which an increase of charges to obtain the 30% of water saving was carried out. In this case, only the *Vol_tot* and *Input* pricing methods present efficient unit values. The *Area* scheme is always less efficient than others.

According to the benchmarking idea, an approach originating in Torgersen et al. (1996), in which an efficient unit is highly ranked if it appears frequently in the reference sets of inefficient decision units, the most frequent water pricing policy is *P1.Vol_Tot*. On the contrary, the policy pricing *P0.Output* does not constitute a 'peer reference' for any other policy.

The relative importance of any input in relation to the final efficiency value is from the weight analysis. This is the case of inefficient policies, where labour is, relatively speaking, the production factor with a greater importance. For instance, the lowest efficiency value is shown in the case of *P1.Area* (0.86374). This means that for the *P1.Area* water policy option, it can reduce the current use of all inputs by 13.626%. In addition, from the assigned weights, it shows that the highest improvement can be obtained in labour. In other words, small decreases in labour use will result in large increases in efficiency. By contrast, the weights assigned to the water are quite lower.

In Table 4, the eco-efficiency value and the optimal DEA weights that take into account environmental and technical concerns simultaneously are shown. This is the novel approach tested in this research.

Table 4- Eco-efficiency score and optimal DEA weights

		Weights	Production input			Desirable output		Undesirable output	
Pricing Policy	**Eco-efficiency**	Land	Labour	Capital	Water	Land cover	Added value	Pesticide Risk	Nitrate surplus
P0.Baseline	1.00000	0.00000	0.00000	0.00000	0.00105	0.00000	0.00152	0.00021	0.02185
P0.Vol_tot	1.00000*	0.00000	0.00000	0.00000	0.00105	0.00000	0.00152	0.00021	0.02185
P0.Input	1.00000	0.00000	0.00000	0.00000	0.00092	0.00000	0.00162	0.00025	0.02390
P0.Output	1.00000	0.00000	0.00000	0.00000	0.00152	0.00000	0.00159	0.00000	0.02583
P0.Area	0.97319	0.00130	0.01374	0.00000	0.00074	0.00158	0.00146	0.00000	0.00000
P1.Baseline	0.99565	0.00141	0.01490	0.00000	0.00080	0.00171	0.00158	0.00000	0.00000
P1.Vol_tot	1.00000	0.00000	0.00000	0.00000	0.00842	0.00000	0.00178	0.00000	0.00000
P1.Input	1.00000	0.00000	0.00000	0.00438	0.00261	0.00000	0,00203	0.00000	0.00000
P1.Output	1.00000	0.00000	0.05238	0.00000	0.00168	0.00082	0.00219	0.00000	0.00000
P1.Area	0.95105	0.00152	0.00882	0.00000	0.00141	0.02652	0.00000	0.00000	0.00000

* Weakly efficient according to the slack value

The findings stress that eco-efficiency reaches the best value for 6 out of the 10 options analysed. According to the *P0* scenario, the current pricing policy *P0.Baseline* and the indirect methods *P0.Input* and *P0.Output* are relatively efficient. The *Vol_tot* scheme is efficient, but it shows a slack for the capital value amounts to 2.23 units. In the case of the *P1* scenario, it should be noted that the *Baseline* scheme becomes relatively inefficient against the *Vol_tot* scheme, which produces an efficient score. Again, the *Area* scheme is always less efficient than the others.

Finally, *P1.Vol_tot*, *P0_Baseline* and *P0.Output* are the DEA reference set for the inefficient pricing options. Although the *Input* scheme is always efficient, it is not a 'peer reference' for the other pricing options.

The comparison of the two efficiency analyses shows how the output scheme presents a difference rank when the environmental concern is included. Although the efficiency rank is almost the same, other differences are also found for the *P1.Area* alternative. Nevertheless, the findings highlight a great difference, mainly for the weights assigned to the input and output.

Nitrate surplus is the most important environmental issue, mainly in the current charge price scenario *P0*. Pesticide Risk seems less important from an environmental perspective for reaching an efficient score. Both items are treated as environmental inputs in our model, and any decrease is desirable. In this sense the eco-efficiency score mainly depends on the *Nitrate surplus* and, as a consequence, a few reductions for that input will result in large improvements in efficiency. By contrast, for Land cover, which is treated as an output, any increase is desirable. In our model, we took into account two desirable outputs, namely, Added Value and Land cover. The DEA weights in Table 5 indicate the relative importance of Land cover as a desirable output.

From the findings of Table 5, it should be noted that the policy options that are inefficient (*P0.Area*, *P1.Baseline* and *P1.Area*) have the highest weight for the input labour. At the same time, a high magnitude for Land cover, which is a favourable output, is shown.

Overall, according to the DEA benchmarking analysis, the *P1.Vol_tot* option has a higher frequency in the DEA reference set for the inefficient pricing options.

In the context of WFD, it can be expected that the direct (i.e., volumetric) systems would be the most efficient, face to the indirect systems. In this sense, the findings show that indirect systems, namely, Input and Output, are also relatively efficient.

6. Concluding remarks

Policymakers require a clear overview of the different outcomes deriving from alternative water management policies, and tools need to be improved for supporting the selection of most suitable measures to specific situations. In areas where agriculture imposes great pressure over the resource, water demand measures need to be tested for a more efficient water use in order to adapt to increased scarcity under the climatic change threat. The study presented here is committed to providing knowledge to support the decision-making process for the selection of water pricing measures for irrigation in the specific context of the Water Framework Directive by going beyond a mere cost-effectiveness approach.

We define an aggregated indicator of efficiency where both environmental performance and technical performance are considered. For this purpose, Data Envelopment Analysis, for which no *a priori* information about weights of objectives is needed, is proposed here. Technical inputs and main environmental externalities (negative and positive both) are simultaneously taken into account.

The findings show a difference of rank between the simpler technical efficiency and the eco-efficiency indicator (in which both technical efficiency and environmental efficiency are included) for the output-based pricing policy. For the rest of the pricing options, the differences are smaller, but the eco-efficiency provides more comprehensive ranking criteria than a simpler technical efficiency score. The findings also show a great difference, mainly for the weights assigned to the input and the output. The weights can be seen as a tool to address the decision-making process.

According to the results of this study, the volumetric pricing methods are the most efficient. However, because indirect methods may be easier to implement, under some circumstances, they might be preferable, being without losses in terms of efficiency. It was found that the pricing based on the value of inputs specific to irrigated crops had a maximum efficiency in any scenario. It should be mentioned, however, that inefficiency seems to still mainly depend on technical rather than environmental concerns (i.e., Pesticide risk and Nitrate surplus). According to the assigned weights through DEA, the highest improvement may be obtained by better labour use. As a consequence, policymakers should pay more attention to labour reforms and not just water policy reforms.

We do not claim that Multi-Criteria Decision Analysis (MCDA) is necessarily better than Cost-Effectiveness Analysis (CEA), as argued by other researchers (e.g., Joubert et al., 1997); rather, we believe that it is to some extent complementary and that in the particular policy problem treated in this paper, it is necessary to broaden the spectrum to the inclusion of a wider range of objectives to include eco-efficiency. This efficiency may be a convenient method for ranking policy hypotheses in the absence of information on stated preferences on specific outcomes, as well as some negative impacts. In addition, the relative importance of a broader range of concerns is provided.

It should be recognised that this analysis was carried out according to the short-term hypothesis. The assumption embedded in this reasoning turned out to provide clues that may enhance the understanding of findings. Moreover, DEA is good at estimating the ‘relative’ efficiency of a policy, but it converges very slowly to ‘absolute’ efficiency. In other words, it can tell you how well you are doing compared to your peers but not compared to a ‘theoretical maximum’.

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Appendix

The simulation of the effects of the different water pricing policies are performed through a multi-agent regional linear programming model (Tisdell, 2001; Berbel and Gutierrez, 2005; Giannoccaro *et al.*, 2008) consisting of a static linear programming model in which farmers are assumed to maximise their profits, subject to the following constraints: i) input endowments (land, water sources and labour); ii) technical aspects (agronomic rotations, labour and the irrigation calendar); and iii) general agricultural policy issues, such as the conditionality of eligibility to the single farm payment under a CAP regime[8]. The decision variables of the model are the crops' activity levels (i.e., crop areas), which determine the utilisation of production inputs, including water and farm income.

The objective of the optimisation model is the maximisation of the regional agricultural net revenue as follows:

$$M \quad \sum_j \lambda_j \left(a\left(\sum_i G x_{ij} \; x_{ij} \right) M + F_j + S_j \; l \right) \qquad (1)$$

s.t.

$$\sum_i x_{ij}\, c_{iz} \le v_{zj} \qquad (2)$$

$$\sum_i x_{ij}\, w_i \le a_j \qquad (3)$$

$$x_{ij} \ge 0 \qquad (4)$$

λ_j = weight of the j farm type;

[8] The single farm payment scheme was introduced by Regulation (EC) No 1782/2003 of 29 September 2003, establishing common rules for direct support schemes under the EU Common Agricultural Policy.

GM_{ij} = gross margin of the crop i on the j farm type;

x_{ij} = activation level (ha) of the i production process, by the j farm type;

c_{iz} = crop technical coefficient for z constraint (except water);

v_{zj} = resource availability for z constraint (except water);

w_i = water consumption for i crop (m^3/ha);

a_j = water availability for j farm;

F_j = fixed running costs of the j farm type;

SFP_j = the single farm payment by farm type, under the CAP regime.

According to the ISTAT (2000) data, labour is provided by the farming family (in 95% of cases) and conducted by elderly farmers (40% of whom are over 65 years old). Farms were classified into three main groups according to farm size and cropping patterns. All of the farmers are assumed to use the same production technology and the same irrigation technique. Drip irrigation is the dominant technique, regardless of the farm size. The major differences between farms concern labour. Small farms' labour is provided by the farmer's family members, while in the case of large farms, the labour is provided by hired workers. The three types of farms also differ in terms of the "single farm payment" under the current CAP regulation. In addition, there are some relevant differences among the crops (such as yields, prices, and input uses), which have been included in the model.

In the technical coefficients are considered the agronomic rotations typically adopted by the farmers in the area. Input and output prices are based on the average (2004-2007) local market prices (Bulletin of the Chamber of Commerce). The size of each farm is fixed. Demand and supply constraints (agronomic operations, input availability, permanent crop area, and CAP framework) according to the farms' features are implemented. The resource constraint for water is specified to accommodate the water delivery schedule from the CBC, which delivers some 106,000,000 m^3 between April and November. In the case of the non-CBC water source, there are constraints with regard to deliverability, and availability is estimated at 89,000,000 m^3 at the most.

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