Instituto Nacional de Investigación y Tecnología Agraria y Alimentaria (INIA)
Available online at www.inia.es/sjar
doi: http://dx.doi.org/10.5424/sjar/20110904-449-10

Spanish Journal of Agricultural Research 2011 9(4), 981-999
ISSN: 1695-971-X
eISSN: 2171-9292

Irrigation water pricing instruments: a sustainability assessment

J. Gallego-Ayala[1], J. A. Gomez-Limon[2]* and M. Arriaza[2]

[1] *Water Research Institute of Mozambique. Av. Patrice Lumumba nº 770. Maputo. Mozambique*

[2] *Instituto de Investigación y Formación Agraria y Pesquera (IFAPA). Junta de Andalucía. Centro Alameda del Obispo. Apdo. 3092. 14080 Córdoba. Spain*

Abstract

The Water Framework Directive (WFD) requires all EU member states to implement water tariffs to recover the costs of water services. This paper analyzes the potential consequences of different instruments for irrigation water pricing (area, volumetric, two-part tariff and block-rate), studying their impact on the sustainability of irrigated areas. The application performed focuses on the Campos district in the Spanish province of Palencia, using simulation models based on positive mathematical programming to simulate farmers' behavior in response to the above-mentioned pricing instruments. A multidimensional set of sustainability indicators (economic, social and environmental) for each instrument is obtained, making it possible to construct a composite indicator for irrigated agriculture (CIIA) in order to measure the overall sustainability performance of irrigated farms. Results show that pricing irrigation water will have a negative impact on overall sustainability, since economic (profitability) and social (generation of employment) sustainability will decline, while only a slight improvement in environmental sustainability will be obtained. However, we show that in order to fulfill WFD requirements, block-rate pricing results in high rates of public-sector revenues derived from irrigation water payments and promotes a significant reduction in the demand for irrigation water with the lowest reductions in farm sustainability measured in terms of the CIIA.

Additional key words: composite indicators; irrigated agriculture; positive mathematical programming; water policy.

Resumen

Tarifación del agua de riego: una evaluación de la sostenibilidad

La Directiva Marco de Aguas (DMA) exige a los estados de la UE la introducción de tarifas para la recuperación de los costes del agua. Este trabajo analiza las consecuencias de la hipotética implementación de diferentes formas de tarifación del agua de riego (por superficie, volumétrica, binómica y por tramos), estudiando su impacto sobre la sostenibilidad de las zonas regables. La aplicación empírica realizada se ha centrado en la Comarca de Campos (Palencia). Con este propósito se han empleado modelos de simulación basados en la programación matemática positiva, los cuales permiten simular el comportamiento productivo de los regantes ante la aplicación de los diferentes instrumentos analizados. La resolución de estos modelos ha permitido obtener, para cada instrumento de tarifación considerado, un conjunto multidimensional de indicadores de sostenibilidad (económicos, sociales y ambientales), a partir de los cuales se ha desarrollado un indicador sintético de sostenibilidad para el regadío (CIIA). Los resultados obtenidos ponen de manifiesto cómo la puesta en funcionamiento de las distintas formas de tarifación generarían un efecto negativo sobre la sostenibilidad del regadío en términos del CIIA, ya que empeorará sus sostenibilidad económica (rentabilidad) y social (generación de empleo), proporcionando tan sólo ligeras mejoras ambientales. En cualquier caso, al objeto de cumplir con la exigencia legal de la DMA, se evidencia que la forma de tarifación por tramos permite obtener elevados niveles de recaudación pública derivados de la tarifación del agua de riego y promueve una reducción significativa de la demanda de agua de riego con las menores reducciones en la sostenibilidad de las explotaciones medida en términos del CIIA.

Palabras clave adicionales: agricultura de regadío; indicadores compuestos; política de aguas; programación matemática positiva.

*Corresponding author: jglimon@uco.es
Received: 17-11-10. Accepted: 15-04-11

Introduction

The growing demand for water in Spain has increased the relative scarcity of this resource. This situation has led to an intense debate about the efficiency of water use in the agricultural sector, which is the main consumer of water (75% of total national withdrawals for economic activities in 2006; INE, 2008). Apparently poor management of this resource in Spanish irrigation (large water losses and application to crops with low profitability and demanding little labor) has served as an argument supporting, as an indispensable solution, the implementation of demand policies typical of a "mature" water economy[1], especially water pricing (Molle and Berkoff, 2007).

The situation of water economy maturity is not unique to Spain, but shared by other states in the European Union (EU), which is why the EU decided to develop a common policy for water management. The result was the approval of Directive 2000/60/EC of the European Parliament and Council, which establishes a framework for community action in the field of water policy (in short, the Water Framework Directive or WFD). The WFD established water pricing as the EU's preferred water-demand control policy (art. 9). The Directive obliges member states to apply tariffs for water use before 2010 in order to provide adequate incentives to use water efficiently. This is expected to contribute to the achievement of the environmental objectives (the "good status" of water bodies) established in the Directive.

Although water pricing is an environmental requirement, the logic on which the instrument is based is purely economic. In this regard, irrigation farmers, according to economic theory, will respond to higher water prices by reducing their consumption, thus relieving the quantitative pressure on water bodies. Pricing affects not only the demand for irrigation water, but has further economic, social and environmental effects. The scientific community has performed extensive studies of the multi-dimensional impacts of pricing irrigation water. Studies pertaining to Spain include those by Varela-Ortega *et al*. (1998), Berbel and Gómez-Limón (2000), Gómez-Limón and Riesgo (2004), Gallego-Ayala and Gómez-Limón (2008), Iglesias and Blanco (2008) and Berbel *et al*. (2009). These studies examine the multi-dimensional impacts of pricing, presenting the performance of politically relevant indicators separately. However, in order to understand the implications of water pricing for the sustainability of irrigated agriculture, it is preferable to examine all its impacts jointly, taking into account the economic, social and environmental implications of this policy instrument simultaneously. This has already been done by Giupponi (2007) and Bartolini *et al*. (2010), using a Decision Support System and a composite index, respectively.

Furthermore, most previous studies consider volumetric pricing as the only means of applying the cost-recovery instrument; it would therefore be of interest to analyze other water pricing instruments.

Within this framework, the objective of the paper is twofold. First, we aim to develop a methodology to analyze *ex-ante* the impacts of alternative instruments of water pricing on the sustainability of irrigated agriculture by estimating a composite sustainability index, following the approach developed by Gómez-Limón and Sanchez-Fernandez (2010). Second, we aim to demonstrate the usefulness of this methodology in a real setting as a tool for technicians and policy makers to support the design and implementation of instruments for better governance of irrigated agricultural systems. For this purpose, the proposed procedure is implemented in the farming district of Campos in the Province of Palencia (Spain).

Abbreviations used: ABAND (risk of farming abandonment); BALN (nitrogen balance); BALP (phosphorus balance); CAP (common agricultural policy); CHD (Confederación Hidrográfica del Duero); CIIA (composite indicator for irrigated agriculture); $CIIA_{eco}$ (composite indicator for irrigated agriculture - economic dimension); $CIIA_{env}$ (composite indicator for irrigated agriculture - environmental dimension); $CIIA_{soc}$ (composite indicator for irrigated agriculture - social dimension); COV (soil covering); EMPLT (farm employment); ENBA (energy balance); EU (European Union); GDP (contribution to the regional GDP); PEST (pesticides risk); PMP (positive mathematical programming); SEAS (seasonal labor); SFP (single farm payment); SPEC (specialization); TGM (total gross margin); WAT (water consumption); WFD (Water Framework Directive).

[1] As stated by Randall (1981), a "mature" water economy is characterized by an "inelasticity supply of 'new' water and the need for expensive rehabilitation of aging projects, more direct and intensive competition among different kinds of users and greatly increased interdependencies among water users".

Material and methods

Case study

We focus on the Campos farming district, located in the centre of the Northern Spanish plateau, in the province of Palencia. Its high altitude (between 700 and 800 m.a.s.l.) and great distance from the sea give it a clearly continental climate, with an average rainfall of around 500 mm per annum, spread heterogeneously over the year. Irrigated agriculture is thus the only alternative to the rain-fed monoculture of winter cereals, allowing the introduction of summer crops.

Irrigated agriculture in the Campos district covers 37,829 hectares, accounting for 14.5% of its utilized agricultural area, which is distributed among 2,096 farms (the average irrigated farm has 18.0 ha).

These areas were transformed into irrigated land in the latter half of the 20th century through regulation systems for the headwaters of its main rivers (Pisuerga and Carrión) and irrigation channels for transporting these surface waters. These infrastructures are publicly owned and are managed by the Basin Authority (*Confederación Hidrográfica del Duero*, CHD). The existing secondary distribution network is also publicly owned, but is managed and maintained by communities of irrigators.

The CHD allocates around 8,000 m^3 ha^{-1} of water a year to communities of irrigators. The water is measured at the entrance to the main irrigation channels. Transport efficiency through the main channels is 80.3%, while distribution efficiency through the secondary network is 85.5% (CHD, 2007). The average annual amount of water that actually reaches the plots is thus about 5,500 m^3 ha^{-1}. Sprinkler and furrow irrigation are of about equal importance.

The most important irrigated crops are winter cereals (wheat –*Triticum aestivum* L.– and barley –*Hordeum vulgare* L.), which occupy 49.8% of the total area, alfalfa –*Medicago sativa* L.– (26.3%), sugar-beet –*Beta vulgaris* L.– (9.9%), maize –*Zea mays* L.– (9.4%) and sunflower –*Helianthus annuus* L.– (4.6%).

Irrigation water pricing is currently based on a fixed quantity, depending on the area irrigated, which in turn comprises three different items. First, there are two tariffs paid to the CHD: a) the *regulation fee*, for management of the infrastructures controlled by this public body (main reservoirs and channels), which is set at € 24 ha^{-1} yr^{-1}; and b) the *water use tariff*, for use of the publicly owned secondary network, ranging between € 10 and € 30 ha^{-1} yr^{-1}, depending on the characteristics of these networks in the individual irrigated areas. On top of all that, communities of irrigators charge *contributions* of between € 20 and € 30 ha^{-1} yr^{-1} to cover operating and maintenance costs. The total fees paid for irrigation water in the study area amount to between € 55 and € 80 ha^{-1} yr^{-1}, equivalent to € 0.015 m^{-3}.

We selected this area because it is representative of Spanish inland irrigated agricultural systems and also because the multifunctional character of irrigated agriculture is evident (Gómez-Limón and Gómez-Ramos, 2007).

Alternatives for irrigation water pricing

As pointed out, the WFD forces member states to apply tariffs for water use in order to provide adequate incentives to use water efficiently and recover costs. However, this European directive does not specify how water pricing should be implemented. We have selected four irrigation water pricing instruments for their potential implementation in the study area, taking into consideration the various alternative methods that are applicable to irrigation water pricing (Tsur and Dinar, 1997; Johansson *et al*., 2002; Easter and Liu, 2005; Molle and Berkoff, 2007) and the specific characteristics of the case study (public irrigated lands and surface water resources). These instruments are:

— *Pricing per unit irrigated area*. Farmers pay for each hectare actually irrigated instead of the current system, which charges for each hectare eligible for irrigation. Eleven pricing scenarios are suggested for the simulation, increasing progressively from € 0 to € 500 ha^{-1} yr^{-1}. These values were selected bearing in mind the current average payment for water services and expected increases due to the WFD (CHD, 2007).

— *Volumetric pricing*. Based on the volume of water used, eleven pricing levels have been selected, ranging from € 0.00 to € 0.10 m^{-3}. These values are regarded as appropriate for the application of the cost-recovery principle required by the WFD (Gómez-Limón and Riesgo, 2004).

— *Two-part tariff system.* This is a combination of the two pricing systems described above, and levies a fixed tariff per hectare actually irrigated and a volumetric tariff on irrigation water. Nine combinations were generated: three fixed tariffs per hectare (€ 50, € 100 and € 150 ha^{-1} yr^{-1}) and three levels of volumetric pricing (€ 0.02, € 0.04, € 0.06 m^{-3}).

— *Block-rate pricing*. This instrument is based on setting differentiated water prices, which increase progressively on the basis of the band or block of water consumption (Bar-Shira *et al*., 2006). We define three blocks of water consumption based on the crop irrigation requirement and irrigation technology (0-3,000, 3,000-6,000 and more than 6,000 $m^3 ha^{-1}$) and, similar to the volumetric pricing described above, four pricing levels were generated (see Table 1).

Obviously, these different water pricing mechanisms would also have a different impact on agricultural performance in the case study analyzed. However, in this paper we seek to compare them using the rationale of cost-recovery. Thus, we aim to ascertain the pricing instrument that yields the most sustainable performance for any cost-recovery level.

Decision-making heterogeneity and cluster analysis

Modeling farming activity at agricultural system level involves problems of aggregation bias (Hazell and Norton, 1986), which can only be avoided if the farms included in the models fulfill strict homogeneity criteria (Day, 1963): technological homogeneity, pecuniary proportionality and institutional proportionality[2].

The irrigated area under consideration is located within a single agricultural county. Hence, bearing in mind climate and soil homogeneity, and technological, institutional and market characteristics (due to the virtual absence of economies of scale in farming activities), the case study area may be regarded as fulfilling the above-mentioned homogeneity criteria. It is thus reasonable to assume similar behavior for all farmers in the study area and to analyze water pricing instruments via a single simulation model with relatively small problems of aggregation bias. However, such homogeneity in producers' behavior rarely exists in the real world. For this reason, in order to minimize aggregation bias in the simulation, we need to classify farmers in terms of homogeneous groups with regard to their crop-mixes (Berbel and Rodríguez, 1998).

In order to apply cluster analysis we surveyed the farmers in the study area. Taking into account the relatively large number of irrigators operating in the Campos district (2,096), we selected a representative sample of 111 farmers using a quota sampling procedure for interviewing. Through the questionnaire designed, relevant information for this study was collected. For clustering purposes, the data regarding crop-mixes[3] were used as classification variables. Cluster analysis was performed using the Euclidean square distance between farmers' crop-mixes and the Ward method as the aggregation criterion.

The simulation technique: positive mathematical programming

Positive mathematical programming (PMP), developed by Howitt (1995), is a mathematical modeling technique based on a calibration system established by a non-linear yield or non-linear cost function that reproduce the same crop-mix distribution as that observed in the real world. Although both approaches (non-lin-

Table 1. Water pricing alternatives for the block-rate system

Block-rate water allowance	Water price (€ m^{-3})			
	Alternative 1	Alternative 2	Alternative 3	Alternative 4
0 – 3,000 $m^3 ha^{-1}$	0.01	0.02	0.03	0.04
3,000 – 6,000 $m^3 ha^{-1}$	0.02	0.04	0.06	0.08
≥ 6,000 $m^3 ha^{-1}$	0.03	0.06	0.09	0.12

[2] If a multi-criteria perspective is being considered, an additional homogeneity requirement emerges in order to avoid aggregation bias; viz., homogeneity related to choice criteria (see Gómez-Limón and Riesgo, 2004). In any case for this study it is assumed that farmers behave as profit maximizers, and thus, the differences observed in crop-mixes among farmers are due to the different production costs faced by each one of them.

[3] Crop-mixed were actually featured bearing in mind the trinomial crop-irrigation_tecnhnology-tillage_technique. Further information about this issue is discussed in the section entitled "Decision variables".

ear yield and cost functions) are equally valid as means of calibrating mathematical models, the method based on cost calibration is more frequently encountered in the literature than that based on yields. On this premise, we employed quadratic cost functions as the mathematical calibration method, since the main differences among farms (see Section "Case study") lie in their cost production structures rather than their yields, the latter being relatively homogeneous due to similar agro-climate conditions.

The standard calibration procedure described by Howitt (1995) is based on three steps. The first consists of building a linear programming model to obtain the dual-value variables for each of the activities (crops) under consideration. These dual-value variables (λ) represent the marginal cost vector and can be interpreted as the marginal opportunity cost of not having the last unit of resource (Howitt, 2006). In the second step, these variables are used to calibrate the cost functions of the individual crops. Finally, in the third step, cost function parameters are used to define a non-linear objective function that will reproduce base-year crop distribution.

The original PMP method has received criticism, some shortcomings of the technique being identified (Heckelei and Britz, 2005; Henry de Frahan *et al.*, 2007). This has led to further development of the PMP with the aim of mitigating the drawbacks of the original approach. Röhm and Dabbert (2003) present an extension of the PMP that permits a greater amount of substitution between similar crops (called "variant activities") than between other less similar crops (activities). The concept of variant activities can thus be applied to either the same crop that is grown using different techniques (*e.g.* irrigated and rain-fed) or crops that belong to the same family (Röhm and Dabbert, 2003). This property is suitable for identifying relevant water pricing scenarios, since farmers would presumably substitute irrigated crops for rain-fed ones.

Taking into account activities i and their possible variants j, the mathematical formulation of the extended PMP that allows obtaining the required dual values is as follows (Röhm and Dabbert, 2003):

$$Max\ TGM = \sum_i \sum_j \left(p_{i,j} \cdot y_{i,j} - c_{i,j} + s_{i,j} \right) x_{i,j} + SFP \quad [1a]$$

subject to:

$$\sum_i \sum_j \left(x_{i,j} \right) \leq \sum_i \sum_j \left(x_{i,j}^0 \right) \quad [\lambda_{land}] \qquad [1b]$$

$$\sum_j \left(x_{i,j} \right) \leq \sum_j \left(x_{i,j}^0 \right) \left(1 + \varepsilon_1 \right) \quad [\lambda_i] \quad \forall i \qquad [1c]$$

$$x_{i,j} \leq x_{i,j}^0 \left(1 + \varepsilon_2 \right) \quad [\lambda_{i,j}] \quad \forall i,j \qquad [1d]$$

$$\varepsilon_2 > \varepsilon_1 \qquad [1e]$$

$$A\vec{X} \leq \vec{B} \qquad [1f]$$

$$x_{i,j} \geq 0 \quad \forall i,j \qquad [1g]$$

Eq. [1a] represents the linear programming model objective function, where TGM is the total gross margin. TGM is calculated as the sum of the gross margins resulting from each activity. The objective function is therefore a function of the area allocated to each crop, $x_{i,j}$ (hectares devoted to crop i, with variant j). These $x_{i,j}$ are the decision variables of the model. To calculate TGM it is also necessary to possess the following technical coefficient data, where $p_{i,j}$, $y_{i,j}$, $c_{i,j}$ and $s_{i,j}$ represent crop prices (measured in € kg^{-1}), yield (kg ha^{-1}), variable cost (€ ha^{-1}) and common agricultural policy (CAP) coupled subsidies per unit area (€ ha^{-1}), respectively. TGM also includes the single farm payment (SFP), which is based on farmers' historical payments measured in euros per hectare[4].

The above-mentioned model has a set of constraints. Eq. [1b] is a structural constraint that limits total agricultural land available, where $x_{i,j}^o$ represents the crop-mix observed in the base year. This constraint produces the dual value of the land (λ_{land}) that was used to intercept the dual value of the least profitable crop. Eq. [1c] and [1d] represent the calibration constraints. Eq. [1c] represents the constraints for total activities, where λ_1 is a small positive number and Eq. [1d] represents the constraints for the variant activity, with λ_2 another small positive number that must satisfy Eq. [1e]. Finally, Eq. [1f] includes the set of "ordinary" constrains (*i.e.*, agronomic, legal, market, etc.) that farmers need to fulfill in this calibration model.

Adding Eqs. [1c] and [1d] forces an optimal solution in the linear programming model that reproduces the

[4] The *Single Farm Payment* (SFP) is a subsidy that agricultural producers receive independently of their crop-mixes or the yields achieved, which is fixed individually on the basis of the amount of subsidies they have been granted in the past. Hence, the SFP should be considered "fixed income", regardless of crop decisions taken by farmers. Therefore, the SFP is just added into Eq. [1a] in order to calculate farmers' TGM.

activities observed in the base year ($x^{o}_{i,j}$). As a result of introducing these two constraints, the model solution generates dual values for the individual activities. Eq. [1c] produces the dual values of activities λ_i and Eq. [1d] the dual values of the variant activity $\lambda_{i,j}$. Nonetheless, since the number of constraints exceeds the number of variables in the primal, some of the variables have dual values equal to zero. This circumstance is observed in the dual value of the least profitable activity (λ_i) and in that of the least profitable variant activity ($\lambda_{i,j}$). However, this situation can be resolved by employing the calibration method developed by Röhm and Dabbert (2003) for the least profitable crop and variant activity.

Once the dual values have been obtained, they are used to calibrate the cost function of the individual activities. These parameters are also used to define the new objective function for the PMP model. Eq. [2] presents the objective function of the extended version of the PMP, including these non-linear cost functions:

$$Max\ TGM = \sum_i \sum_j \left\{ x_{i,j} \left[y_{i,j} p_{i,j} - c_{i,j} \left(\alpha_{i,j} + \beta_{i,j} x_{i,j} + \gamma_{i,j} \sum_j x_{i,j} \right) + s_{i,j} \right] \right\} + SFP \quad [2]$$

where $\alpha_{i,j}$ (the axis intercepted coefficient), $\beta_{i,j}$ (the slope coefficient of variant activity level) and $\gamma_{i,j}$ (the slope coefficient of total crop activity level) denote the cost function parameters with the following mathematical expressions:

$$\alpha_{i,j} = 1 - \frac{\lambda_i + \lambda_{i,j}}{c_{i,j}};\ \beta_{i,j} = \frac{\lambda_{i,j}}{c_{i,j} x^0_{i,j}};\ \gamma_{i,j} = \frac{\lambda_i}{c_{i,j} \sum_j x^0_{i,j}} \quad [3]$$

This PMP approach as a simulation model has already been followed by Cortignani and Severini (2009), Gallego-Ayala and Gómez-Limón (2009) and Henseler *et al*. (2009) among others.

Calculating the composite sustainability indicator

The following sections explain how we calculated the Composite Indicator for Irrigated Agriculture (CIIA) used in this study according to the guidelines suggested by the OECD-JRC (2008).

First, we select a set of indicators to assess farm sustainability covering the economic, social and environmental components of sustainability. We opt for the theoretical framework proposed by van Cauwenbergh *et al*. (2007), known as SAFE (Sustainability Assessment of Farming and the Environment Framework). This approach considers the goods and services provided by agricultural ecosystems, resulting in the primary level of the hierarchy; *i.e.* the "principles" that are correlated with the three dimensions of sustainability: economic, social and environmental. From these principles we can derive the "criteria" that comprise the second order of the hierarchy, from which we finally derive the "indicators" themselves. This framework has been the starting point for our selection of

Table 2. Selected basic indicators

Sustainability dimensions	Indicators	Measurement units
Economic	Total Gross Margin (*TGM*)	€ ha^{-1}
	Contribution to the regional GDP (*GDP*)	€ ha^{-1}
Social	Farm employment (*EMP*)	Working-hours ha^{-1}
	Seasonal labor (*SEAS*)	%
	Risk of farming abandonment (*ABAN*)	%
Environmental	Specialization (*SPEC*)	%
	Soil covering (*COV*)	%
	Nitrogen balance (*BALN*)	kg N ha^{-1}
	Phosphorus balance (*BALP*)	kg P ha^{-1}
	Pesticides risk (*PEST*)	kg ha^{-1}
	Water consumption (*WAT*)	m^3 ha^{-1}
	Energy balance (*ENBA*)	kcal ha^{-1}

indicators. Table 2 displays those finally chosen. For further information about the interpretation of each one of these indicators, interested readers may consult the research by Gómez-Limón and Sanchez-Fernandez (2010).

Second, as indicators chosen are measured in different units, a normalization procedure is required before operating with them. Regarding the normalization techniques available (Freudenberg, 2003), we employ "min-max" normalization, making values of all normalized indicators vary within a dimensionless range (0,1), where 0 represents the worst possible value of the indicator (*i.e.,* the least sustainable value of all the alternative water pricing scenarios) and 1 the best (*i.e.,* the most sustainable value among the alternative water pricing scenarios).

Thirdly, we also require an indicator weighting in order to differentiate their relative importance. Regarding this point, it is worth mentioning the research by Gómez-Limón and Atance (2004), who addressed the relative importance of public objectives that should guide agricultural policy in the Spanish region of Castilla y León, where the case study is located. In order to achieve this objective, they applied the Analytic Hierarchy Process to a hierarchical structure in which criteria were identified with economic, social and environmental objectives. In order to estimate the weights assigned to each generic objective they carried out a survey of citizens, obtaining a sample of 321 valid questionnaires (pair comparisons to build individual Saaty matrices). Because of the similarity between the generic objectives considered in that study and the three basic dimensions of sustainability analyzed in this paper, we regard the results obtained by Gómez-Limón and Atance (2004) as suitable for weighting our criteria. Thus, we have: w_{eco}=28.5%, w_{soc}=39.9% and w_{env}=31.7% (see second row in Figure 1).

For the case of sub-criteria (indicators) weighting, we employed a panel of 16 experts from universities and research institutes. Each of these experts completed three Analytic Hierarchy Process questionnaires in order to provide the pair-wise comparisons needed to calculate relative weightings for the various indicators considered in each of the three dimensions of sustainability (economic, social and environmental). Once each expert's weightings had been estimated, these weights were aggregated using the geometric mean. This provided a technical consensus regarding relative weightings for indicators within each dimension of sustainability, as can be seen in the third row of Figure 1. In any case, to make such weights operative, it is necessary to normalize them (normalized weights should add up to one). In order to meet this requirement, the weight of each sub-criterion (indicator) is multiplied by the weight of its own criterion (importance of economic, social or environmental sustainability). The final results can be seen in the fourth row of Figure 1.

Fourth, we selected the functional form of aggregation. This step is one of the most controversial when building composite indicators (Morse *et al*., 2001; Ebert and Welsch, 2004; Hueting and Reijnders, 2004; Böhringer and Jochem, 2007). The choice is not trivial, as it influences the type of compensation or "mar-

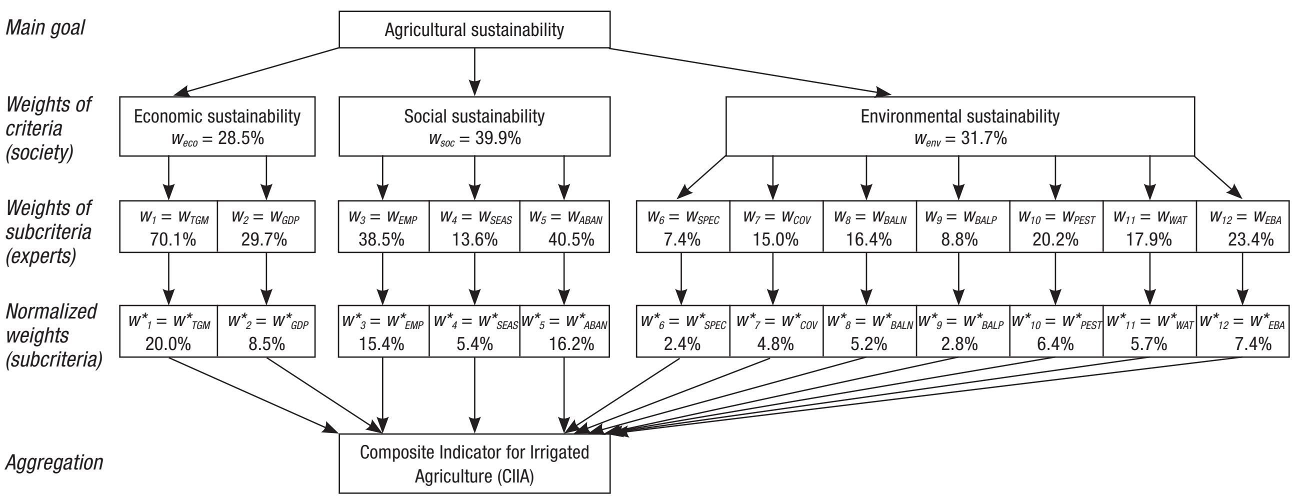

Figure 1. Weights for sustainability dimensions and basic indicators.

ginal rate of substitution" among indicators (Munda, 2005), depending on the algebraic alternative. In brief, additive linear functions implicitly assume total compensation among indicators and multiplicative and geometric functions permit partial compensation; conversely, there are also non-compensatory multi-criterion functions. Thus, depending on the aggregation method employed to obtain the indices, the results and the conclusions drawn from them may differ from case to case. We address the above challenge by aggregating them as a weighted sum of their normalized values, a procedure frequently used in assessing agricultural sustainability (Sands and Podmore, 2000; Rigby *et al*., 2001; Hajkowicz, 2006; van Calker *et al*., 2006; Qiu *et al*., 2007). Following this aggregation approach, the overall composite indicator for irrigated agriculture was obtained as follows:

$$CIIA = \sum_{k=1}^{k=12} w^*_k \cdot I_k \qquad [4]$$

where w^*_k is the normalized weight associated with indicator k, and I_k is the normalized value of indicator k.

Partial composite indicators were also obtained for each sustainability dimension (economic, social and environmental) to analyze their relative importance, using the following expressions:

$$CIIA_{economic} = \sum_{k=1}^{k=2} w^*_k \cdot I_k \qquad [5]$$

$$CIIA_{social} = \sum_{k=3}^{k=5} w^*_k \cdot I_k \qquad [6]$$

$$CIIA_{environmental} = \sum_{k=6}^{k=12} w^*_k \cdot I_k \qquad [7]$$

Source of input information for models and indicators

The information needed as inputs for the simulation models, as well as for calculating the sustainability indicators, was collected from both primary and secondary sources.

The primary data source was the previously mentioned survey of a representative sample of irrigators. This survey allowed us to collect information about the socioeconomic characteristics of the owners, the structural characteristics of their farms, their crop plans and the agricultural practices and techniques they employed.

The information collected enabled us to characterize the diversity of farms in the area and to establish the farm-types in the district. The survey was also the main source of information used to construct the simulation models and calculate the base indicators.

Secondary information was also collected for output and input prices, coefficients of nitrogen, phosphorus and the energy content of inputs and outputs, water requirements for irrigated crops, efficiency of irrigation systems and pesticide toxicity. These data are identical for all the producers in the area and were therefore used to feed the models and to calculate the base indicators in every case.

Modeling

Defining irrigated farm-types in the study area

Cluster analysis resulted in four homogeneous groups of farmers. Table 3 shows the main features of these farm-types resulting from statistically significant differences among clusters, characterized by their crop plans, as well as the tillage and irrigation techniques employed.

The farm-types were the basic units of analysis used to construct the simulation models in order to minimize the aggregation bias discussed in the section entitled "Decision-making heterogeneity and cluster analysis". A different model was constructed for each group of farmers in order to independently simulate the effects of irrigation water pricing methods. The results obtained for each group of producers were subsequently aggregated at district level by weighting the sum of the results for each farm-type, based on the area represented by each of them. Both farm-type and aggregated results are helpful for policy makers when designing and implementing pricing instruments.

Decision variables

The areas devoted to each of the most common crops in the study area ($x_{i,j}$) were the decision variables used to build the simulation models. Due to the differences in cost and existing yields, we found it most appropriate to characterize these activities on the basis of three factors: crop, irrigation technology and tillage tech-

Table 3. Main features of different farm-types in the Campos district

	Minimum tillage cereal growers	Conventional farmers with diversified production	Min. tillage and sprinkler irrigation cereal-sugar-beet growers	Direct sowing cereal growers
Percentage of farmers sampled	37.8	19.8	31.5	10.8
Percentage of total area analyzed	41.5	17.5	29.4	11.6
Farm size (ha)	125	101	105	122
Percentage of irrigated land	31	42	48	40
Age (years)	44.5	44.2	46.1	43.5
Main tillage technology	Minimum tillage (75% of the total farm area)	Conventional tillage (100% of the total farm area)	Minimum tillage (74% of the total farm area)	Direct sowing (80% of the total farm area)
Main irrigation technology	Furrow (62% of the irrigated area)	Furrow (59% of the irrigated area)	Sprinkler (66% of the irrigated area)	Furrow (71% of the irrigated area)
Main irrigated crops	Winter cereals (51%) and alfalfa (26%)	Winter cereals (46%), alfalfa (23%), sunflower (12%) and maize (10%)	Winter cereals (44%), alfalfa (30%) and sugar-beet (12%)	Winter cereals (70%)

niques. We considered only the combinations actually used by the farmers in the area, selecting a total of 34 decision variables, as shown in Table 4.

The models simulate the productive behavior of the set of farm-types analyzed, including both their irrigated areas, whether these were used for irrigated or rain-fed crops, and those which are purely rain-fed (with no possibility of irrigation). This is a novelty in the literature and may prove to be more suitable for simulating irrigation farmers' behavior.

Different groups of activities were defined following the extension of the PMP developed by Röhm and Dabbert (2003) to make the models more flexible and to allow for a greater degree of substitution among individual activities (irrigated crops for rain-fed crops) in the face of water pricing policy. First, the activities *i* were established on the basis of crop and tillage technology alternatives (*e.g.* wheat-conventional_tillage, sunflower-minimum_tillage, etc.). Second, for each activity *i*, variant activities *j* were defined on the basis of the irrigation techniques available for each activity (irrigated_furrow, irrigated_sprinkler and rain-fed). The resulting variant activities comprise a package of "crop-tillage_technology-irrigation_technology" (*e.g.,* wheat-conventional_tillage-rain-fed, wheat-conventional_tillage-sprinkle_irrigation). Finally, the different variant activities of winter cereals (wheat, barley and oats) were combined, due to the botanical similarity of these crops in one single group of variant activities. This assumed that within any group of these variants, there would be a greater degree of substitution (one irrigated crop for the same crop in rain-fed conditions, or one winter cereal for another crop from this botanic family) than between other activities (one crop with a specific tillage technology for another crop or another tillage system).

Modeling of pricing per unit of irrigated area

The model built to simulate the implementation of a fixed fee over the irrigated area appears below:

$$Max \sum_i \sum_j \left\{ x_{i,j} \left[y_{i,j} p_{i,j} - c_{i,j} \left(\alpha_{i,j} + \beta_{i,j} x_{i,j} + \gamma_{i,j} \sum_j x_{i,j} \right) + \right. \right.$$

$$\left. \left. + \; s_{i,j} - t_s \cdot \sum_{i^r} \sum_{j^r} x_{i,j} \right] \right\} + SFP \qquad [8a]$$

Table 4. Decision variables for the study area

Irrigation technology Tillage technology*	Furrow			Sprinkler			Rain-fed		
	CT	MT	DS	CT	MT	DS	CT	MT	DS
Wheat	X	X	X	X	X		X	X	X
Barley	X	X	X	X	X	X	X	X	X
Oat							X	X	
Green peas								X	X
Sunflower	X						X	X	
Alfalfa	X	X		X	X		X		
Grain maize	X								
Green maize	X			X					
Sugar-beet				X	X				

* CT: conventional tillage; MT: minimum tillage; DS: direct sowing.

Subject to:

Total area constraint:

$$\sum_i \sum_j x_{i,j} \leq AREA \qquad [8b]$$

Total irrigated area constraint:

$$\sum_{i^r} \sum_{j^r} x_{i,j} \leq AREA_{irrigated} \qquad [8c]$$

Sprinkler irrigation area constraint:

$$\sum_{i^{rs}} \sum_{j^{rs}} x_{i,j} \leq AREA_{sprinkler} \qquad [8d]$$

Water availability constraint:

$$AREA \cdot ALLOT \geq \sum_{i^r} \sum_{j^r} x_{i,j} \cdot \frac{WR_{i,j}}{Efic_{i,j}} \qquad [8e]$$

Labor requirements constraint:

$$\sum_i \sum_j l^p_{i,j} \cdot x_{i,j} \leq L^p \quad \forall p \qquad [8f]$$

Alfalfa rotation constraint:

$$\sum_j x_{alf-CT,j} + \sum_j x_{alf-MT,j} \leq 0.55 \times AREA \qquad [8g]$$

Sugar-beet CAP constraint:

$$x_{sug-CT} + x_{sug-MT} \leq 50\% \frac{\text{Sugar beet quota}}{\text{Sugar beet yield}} \qquad [8h]$$

Alfalfa market constraint:

$$\sum_j y_{alf-CT,j} x_{alf-CT,j} + $$

$$+ \sum_j y_{alf-MT,j} x_{alf-MT,j} \leq \text{Production}_{time\ series\ maximum} \qquad [8i]$$

Non-negativity constraint:

$$x_{i,j} \geq 0 \quad \forall i, j \qquad [8j]$$

Equation [8a] represents the objective function, which is fitted to expression [2]. This includes the fee per irrigated area t_s that would be charged for the crops and variants actually irrigated (indicated by sub-indices i^r and j^r). The parameter of this fee was set to take values ranging from € 0 ha^{-1} to € 500 ha^{-1}.

The first constraint [8b] limits the crop area to the total area (irrigated plus rain-fed) actually available on the farm (*AREA*). Constraint [8c] limits the irrigated area to the available irrigated area ($AREA_{irrigated}$). As this is a short and medium-term model, rain-fed land cannot be converted into irrigated land. For the same reason, the possibility of introducing innovations in irrigation technology was not included. The area using sprinkler irrigation (crops and variants indicated by indices i^{rs} and j^{rs}) is therefore limited to the area currently under irrigation using that technique ($AREA_{sprinkler}$), as established in expression [8d]. Equation [8e] limits the water available for irrigation, where *ALLOT* is the annual water allotment assigned to each farm measured in m^3 ha^{-1}, $WR_{i,j}$ are the water requirements of the crop *i,j* and $Efic_{i,j}$ is the technical efficiency associated with the irrigation technique used for that crop. Constraint [8f] limits the availability of labor during the most critical (i.e. the most labor-intensive) periods of the year (p), with $l^p_{i,j}$ being the labor requirement of crop *i,j* in period p, and L^p the total availability of this input in the same period.

Constraint [8g] was included so that the optimum crop plans resulting from the model would respect the agronomic restrictions on alfalfa growing (see Foltz *et al.*, 1992). Expression [8h] was incorporated in order to permit a suitable simulation to be performed of the restructuring of the sugar-beet market following the latest reform of the Common Market Organization for sugar. In accordance with this reform, sugar-beet growers are obliged to abandon 50% of the production of this crop from the 2008/2009 season, for which they were compensated with € 40 per ton delivered on average during the four-year period 2004-2008 (a quantity which is included, duly annualized, in the SFP). Finally, expression [8i] is the market constraint relating to alfalfa. This constraint is included due to the inflexible demand for this crop caused by the (almost) fixed requirements for feeding livestock (their size is fairly constant as a result of the CAP quotas). Demand for alfalfa is therefore unlikely to exceed the maximum production of the past ten years.

The set of constraints [8b], [8c], [8d], [8e], [8f], [8g], [8h], [8i] and [8j], referred to hereafter as $A\vec{X} \le \vec{B}$, have been included in all calibration and simulation models[5].

Modeling of volumetric pricing

The model for simulating the implementation of volumetric water pricing is as follows:

$$Max \sum_i \sum_j \left\{ x_{i,j} \left[y_{i,j} p_{i,j} - c_{i,j} \left(\alpha_{i,j} + \beta_{i,j} x_{i,j} + \gamma_{i,j} \sum_j x_{i,j} \right) + s_{i,j} - t_w \cdot \frac{WR_{i,j}}{Efic_{i,j}} \right] \right\} + SFP \qquad [9a]$$

subject to:

General constraint: $A\vec{X} \le \vec{B}$ [9b]

where t_w is the volumetric irrigation water tariff, a parameter that can range from € 0.00 to € 0.10 m^{-3}.

Modeling of the two-part tariff system

We simulated the application of a two-part irrigation water pricing system with the following model:

$$Max \sum_i \sum_j \left\{ x_{i,j} \left[y_{i,j} p_{i,j} - c_{i,j} \left(\alpha_{i,j} + \beta_{i,j} x_{i,j} + \gamma_{i,j} \sum_j x_{i,j} \right) + s_{i,j} - t_w \cdot \frac{WR_{i,j}}{Efic_{i,j}} - t_s \cdot \sum_{i^r} \sum_{j^r} x_{i,j} \right] \right\} + SFP \qquad [10a]$$

subject to:

General constraint: $A\vec{X} \le \vec{B}$ [10b]

In this case the values of t_s range from € 50 to € 150 ha^{-1}, and t_w takes values of between € 0.00 and € 0.06 m^{-3}.

Modeling of block-rate pricing

Equation [11a] defines the objective function used to simulate the productive behavior of the farm-types under a system of block-rate pricing for irrigation water:

$$Max \sum_i \sum_j \left\{ x_{i,j} \left[\begin{array}{l} y_{i,j} p_{i,j} - c_{i,j} \left(\alpha_{i,j} + \beta_{i,j} x_{i,j} + \gamma_{i,j} \sum_j x_{i,j} \right) + s_{i,j} - \\ - \left[t_{w1} \cdot (3{,}000 - \phi) + t_{w2} \cdot (\gamma - \eta) + t_{w3} \cdot \varphi \right] \end{array} \right] \right\} + SFP \qquad [11a]$$

subject to:

General constraint: $A\vec{X} \le \vec{B}$ [11b]

First block water constraint:

$$\frac{\sum_{i^r} \sum_{j^r} \frac{WR_{i,j}}{Efic_{i,j}}}{\sum_{i^r} \sum_{j^r} x_{i,j}} + \phi - \gamma = 3{,}000 \qquad [11c]$$

Second block water constraint:

$$\frac{\sum_{i^r} \sum_{j^r} \frac{WR_{i,j}}{Efic_{i,j}}}{\sum_{i^r} \sum_{j^r} x_{i,j}} + \eta - \varphi = 6{,}000 \qquad [11d]$$

[5] Constraint [8h] was not used in the calibration model, bearing in mind that the compulsory abandonment of 50% of sugar-beet production came into force after the farming year 2007/2008 (Baseline scenario –calibration year).

Non-negativity constraints:

$$\phi \geq 0\ ;\ \gamma \geq 0\ ;\ \eta \geq 0\ ;\ \varphi \geq 0 \qquad [11e]$$

where t_{w1} (t_{w1} = € 0.01, € 0.02, € 0.03 and € 0.04 m^{-3}) refers to the unitary water tariff for the first block of water consumption applied to the volume 3,000-ϕ m^3 ha^{-1} derived from expression [11c], t_{w2} (t_{w2} = € 0.02, € 0.04, € 0.06 and € 0.08 m^{-3}) is the second block of water consumption, which charges the quantity $\gamma - \eta$, deduced from expressions [11c] and [11d], and t_{w3} (t_{w3} = € 0.03, € 0.06, € 0.09 and € 0.12 m^{-3}) is the tariff corresponding to the third block of water consumption, which is charged for the volume φ, derived from expression [11d].

Calibration and validation of the models

The PMP models were calibrated taking into account the conditions faced by farmers in the 2007-2008 farming year (*Baseline scenario*). The regulatory framework of the CAP for this period is the Mid-Term Review, which was approved in 2003. This scenario is characterized by the partial decoupling of the CAP payments, which means that producers of arable crops receive a ‘coupled’ area payment (based on the theoretical yields at county level) equal to 25% of the support received as part of the previous CAP scheme (Agenda 2000). The remaining 75% of this support is transferred through a “decoupled” payment (SFP) which is received annually by farmers regardless of their crop-mix. With respect to irrigation, the pricing mechanism in the *Baseline scenario* is the current one as described in the section entitled “Case study”. Taking this institutional framework into account, models for the *Baseline scenario* were calibrated using the information gathered from a survey carried out at farm level in 2008 (see the section entitled “Source of input information for models and indicators”), where all data (crop plans, yields, output and input prices, subsidies and other technical data such as agricultural practices and techniques) refer to the 2007-2008 agricultural season.

As Heckelei and Britz (2005) and Henry de Frahan *et al*. (2007) point out, using observations from a single year, as in this study, may lead to less robust simulation models, as the estimation and inference built under such limited information can produce inconsistent model parameters. As the PMP approach is very sensitive to baseline conditions (it affects the shape of the cost functions to be considered within the objective function in the simulation models), the validation of the models, in terms of the consistency of the parameters recovered, is a crucial part of the methodology employed to assess farmers’ responses to changes in the institutional framework.

In order to check the robustness of the simulation models, we performed an *ex-post* analysis to validate them. For this purpose we checked, for each PMP farm-type model, whether the values of the parameters previously estimated in the calibration step (farming year 2007-2008) can accurately reproduce the crop plans of these producers when they faced a different institutional framework. This was done for the former CAP scenario based on the Agenda 2000 Reform (farming year 2004-2005), which was characterized by public-sector support through direct payments per unit area, calculated taking into account theoretical yields at county level. The Finger-Kreinin similarity index (FK index, see Finger and Kreinin, 1979) was calculated to compare the crop-mix observed in 2004-2005 to that obtained from the simulation in order to quantify the validity of the calibrated PMP models. The FK index ranges from 0 to 100, the latter representing no difference between observed and predicted data (for further details see Blanco *et al*., 2008). Table 5 presents the results obtained for this analytical exercise.

For the four farm-types that we evaluated, FK values exceeded 90%. This supports the validity of the parameters estimated to simulate the farmers’ decision-making in the short- and medium-term and, more specifically, to assess their responses to alternative pricing mechanisms and/or the CAP framework.

Results

Composite indicator for irrigated agriculture

Preliminary results were obtained for each farm-type by running the models explained above. Subsequently, through weighted aggregation of these partial results, we obtained the results for the whole irrigated system, as shown in Tables 6 and 7. Table 6 displays the values of the base indicators for the study area as a whole in each of the simulated pricing scenarios. The overall results are presented in terms of the values obtained for the CIIA composite indicator and its dimensional components ($CIIA_{eco}$, $CIIA_{soc}$ and $CIIA_{env}$), as shown in Table 7.

Table 5. Model validation: observed crop-mix distribution in 2004-2005 vs. simulated crop-mix distribution for the Agenda 2000 scenario

Crops	Minimum tillage cereal growers			Conventional farmers with diversified production			Min. tillage and sprinkler irrigation cereal-sugar-beet growers			Direct sowing cereal growers		
	Observed crops	Simulated crops	FK index	Observed crops	Simulated crops	FK index	Observed crops	Simulated crops	FK index	Observed crops	Simulated crops	FK index
Irrigated winter cereals	16.15	14.88	14.88	19.91	19.07	19.07	19.95	18.89	18.89	24.50	27.91	24.50
Rain-fed winter cereals	60.44	60.60	60.44	51.98	59.18	51.98	40.85	42.68	40.85	51.17	55.27	51.70
Green peas	2.62	4.53	2.62	0.00	0.00	0.00	2.38	5.06	2.38	2.75	1.65	1.65
Irrigated sunflower	0.70	0.78	0.70	4.90	3.87	3.87	0.86	0.83	0.83	1.67	1.17	1.17
Rain-fed sunflower	2.88	3.00	2.88	1.63	2.31	1.63	4.17	4.07	4.07	5.33	3.08	3.08
Irrigated alfalfa	8.04	9.40	8.04	8.32	9.72	8.32	14.34	15.69	14.34	5.33	4.92	4.92
Rain-fed alfalfa	4.21	0.88	0.88	4.39	1.65	1.65	3.04	0.18	0.18	0.75	0.00	0.00
Grain maize	2.17	1.93	1.93	4.00	1.62	1.62	4.50	3.27	3.27	0.00	0.00	0.00
Green maize	0.69	1.26	0.69	0.00	0.00	0.00	2.80	3.80	2.80	0.83	0.98	0.83
Sugar-beet	2.76	2.76	2.76	4.88	3.99	3.99	5.54	5.52	5.52	7.67	5.05	5.05
FK similarity index			95.83			92.13			93.13			92.37

The recent change in the CAP (implementation of the Health Check reform) would have in itself only a slightly negative effect on the sustainability of irrigation farming in the study area. This negative impact is expressed by the decrease in the composite sustainability indicator, which falls from 0.68 (*Baseline scenario*) to 0.61 (see Table 7). This fall is due to the total decoupling of production support, which encourages

Table 6. Basic indicators for the Campos district*

Water pricing scenarios	Economic indicators		Social indicators			Environmental indicators						
	TGM (€ ha^{-1})	GDP (€ ha^{-1})	EMP (Whours ha^{-1})	SEAS (%)	ABAND (%)	SPEC (%)	COV (%)	BALN (kg N ha^{-1})	BALP (kg P ha^{-1})	PEST (kg ha^{-1})	WAT (m^3 ha^{-1})	ENBA (kcal ha^{-1})
Baseline scenario	481.70	270.04	11.51	0.52	0.01	29.77	0.74	33.68	28.59	686.36	6,577	$1.16 \cdot 10^7$
Health Check	456.13	233.83	11.20	0.53	0.05	31.85	0.74	31.40	29.64	589.72	5,998	$1.06 \cdot 10^7$
Pricing instrument: irrigated area												
100 € ha^{-1}	417.53	290.96	10.65	0.56	0.13	35.05	0.74	30.80	30.06	549.90	5,099	$1.00 \cdot 10^7$
300 € ha^{-1}	369.00	329.07	9.03	0.65	0.23	41.54	0.73	31.58	30.98	462.11	2,864	$9.34 \cdot 10^6$
500 € ha^{-1}	343.16	340.15	8.37	0.69	0.29	44.56	0.72	31.34	31.44	419.79	1,920	$8.77 \cdot 10^6$
Pricing instrument: volumetric tariff												
0.01 € m^{-3}	429.30	274.34	10.61	0.56	0.11	34.02	0.74	31.87	30.02	555.69	5,145	$1.03 \cdot 10^7$
0.05 € m^{-3}	375.86	302.79	8.48	0.69	0.22	41.38	0.72	33.76	31.28	439.87	2,282	$9.43 \cdot 10^6$
0.10 € m^{-3}	344.14	350.74	8.26	0.71	0.29	43.83	0.72	32.49	31.66	408.82	1,821	$8.81 \cdot 10^6$
Pricing instrument: two-part tariff system												
50 € ha^{-1} + 0.02 € m^{-3}	397.70	308.49	9.51	0.62	0.17	37.90	0.73	32.39	30.53	498.82	3,662	$9.89 \cdot 10^6$
50 € ha^{-1} + 0.06 € m^{-3}	364.22	340.69	8.57	0.69	0.24	41.82	0.72	32.54	31.32	431.83	2,306	$9.05 \cdot 10^6$
150 € ha^{-1} + 0.02 € m^{-3}	375.98	318.85	8.86	0.66	0.22	40.90	0.72	32.49	30.96	460.20	2,732	$9.51 \cdot 10^6$
150 € ha^{-1} + 0.06 € m^{-3}	350.27	350.61	8.39	0.69	0.27	43.30	0.72	32.03	31.45	419.89	2,011	$8.87 \cdot 10^6$
Pricing instrument: block-rate system												
0.01-0.02-0.03 € m^{-3}	424.46	310.45	9.88	0.60	0.12	35.00	0.73	34.48	30.23	531.77	4,268	$1.06 \cdot 10^7$
0.04-0.08-0.12 € m^{-3}	362.86	449.95	8.63	0.69	0.25	39.85	0.72	35.48	31.27	445.04	2,498	$9.55 \cdot 10^6$

* TGM: total gross margin, GDP: contribution to the regional GDP, EMP: farm employment, SEAS: seasonal labor, ABAND: risk of farming abandonment, SPEC: specialization, COV: soil covering, BALN: nitrogen balance, BALP: phosphorus balance, PEST: pesticides risk, WAT: water consumption, ENBA: energy balance.

Table 7. Simulation results for the Campos district

Water pricing scenarios	Composite indicators*			
	$CIIA_{eco}$	$CIIA_{soc}$	$CIIA_{env}$	CIIA
Baseline scenario	0.19	0.32	0.17	0.68
Health check	0.15	0.29	0.17	0.61
Pricing instrument: irrigated area				
100 € ha^{-1}	0.12	0.23	0.17	0.52
300 € ha^{-1}	0.08	0.11	0.18	0.37
500 € ha^{-1}	0.05	0.05	0.18	0.28
Pricing instrument: volumetric tariff				
0.01 € m^{-3}	0.13	0.24	0.17	0.54
0.05 € m^{-3}	0.08	0.09	0.18	0.35
0.10 € m^{-3}	0.06	0.04	0.18	0.28
Pricing instrument: two-part tariff system				
50 € ha^{-1} + 0.02 € m^{-3}	0.11	0.16	0.18	0.44
50 € ha^{-1} + 0.06 € m^{-3}	0.08	0.08	0.18	0.34
150 € ha^{-1} + 0.02 € m^{-3}	0.08	0.11	0.18	0.37
150 € ha^{-1} + 0.06 € m^{-3}	0.06	0.06	0.18	0.30
Pricing instrument: block-rate system				
0.01-0.02-0.03 € m^{-3}	0.14	0.20	0.18	0.52
0.04-0.08-0.12 € m^{-3}	0.11	0.08	0.18	0.36

* $CIIA_{eco}$: composite indicator for irrigated agriculture - economic dimension, $CIIA_{soc}$: composite indicator for irrigated agriculture - social dimension, $CIIA_{env}$: composite indicator for irrigated agriculture - environmental dimension, CIIA: composite indicator for irrigated agriculture.

greater extensification of production (introduction of rain-fed crops in irrigated areas), which in turn reduces the generation of added value and worsens the social performance of farming (observe the performance of $CIIA_{eco}$ and $CIIA_{soc}$ in Table 7), without introducing significant variations in environmental sustainability ($CIIA_{env}$).

The results obtained for the implementation of irrigation water pricing point in the same direction: the different instruments generate a reduction in the overall sustainability (CIIA index) of the irrigated agricultural system analyzed. The explanation for this lies in the fact that farmers introduce new changes in their productive strategies as a response to water pricing, substituting irrigated crops for rain-fed alternatives (production extensification). This leads to a worsening in the economic and social performance of farming, while environmental sustainability remains practically unchanged.

These economic instruments erode the economic sustainability (performance of $CIIA_{eco}$) of irrigated agriculture, as shown in Table 7. This is mainly because water pricing entails a significant reduction in farmers' private profitability (TGM indicator), produced both by the payments they must make to the authorities in the form of fees (transfer of income from the private to the public sector) and also to changes in their crop plans (irrigated crops substituted by other rain-fed land crops with lower added value).

In addition, the pricing instruments that take into account the actual consumption of irrigation water (volumetric and block-rate pricing, which transmit the cost signal for water use more directly) lead to a smaller reduction in the $CIIA_{eco}$ index than tools that price water regardless of actual consumption (pricing by irrigated area). These differences can be explained because volumetric pricing and block-rate pricing minimize the emergence of the above-mentioned economic inefficiency (reduction of farm profitability due to the introduction of crops with lower added value), leading only a transfer of income from farmers to the public sector. However, it must be stated that this is only true for low volumetric tariffs. As the amount of the charge increases, farmers' income losses grow to a greater extent than the rise in public sector revenues (inefficiency also increases).

The introduction of water pricing would similarly damage the social sustainability of irrigated land in the study area. We can now see how the values of the $CIIA_{soc}$ index fall rapidly and significantly as the charges for irrigation water increase. This is due to three causes: a) the loss of direct employment in irrigation (reduction in the EMP indicator), due to a downturn in demand for work as a consequence of substituting irrigated crops for less labor-intensive rain-fed crops, b) an increase in the seasonality of labor demand (SEAS indicator), caused by the change in crop plans, and c) an increase in the risk of farm abandonment (ABAN indicator), which is closely linked to the loss of farm income for the reasons mentioned above. In any case, unlike the $CIIA_{eco}$ index, the development of the social component of sustainability does not display significant differences as a result of the pricing instrument used; they all generate significant decreases in the $CIIA_{soc}$ index.

From the point of view of environmental sustainability ($CIIA_{env}$), the implementation of irrigation water pricing is almost neutral (see Table 7). These results can be considered counterintuitive at first glance. However, they are a reflection of how the positive and negative environmental externalities associated with irrigation water pricing offset each other, in terms of $CIIA_{env}$ index, generating an improvement in the WAT base indicators (reduction in the pressure of irrigation on water resources) and PEST (reduction in the release of plant protection products to the environment). However, at the same time, the SPEC indicator (increase in single-crop farming of winter cereals on non-irrigated land - reduction in biodiversity), COV (increase in the risk of wind and water soil erosion), BALN and BALP (increase in diffuse contamination by nitrogen and phosphorus derived from farming)[6] and ENBA (reduction in the efficiency of the irrigation system as a CO_2 sink) all record worse scores. This mixed environmental effect of pricing means that once the different base indicators have been weighted, the $CIIA_{env}$ index remains relatively stable when this type of water tariff is applied.

Finally, Table 8 presents the impact of the different scenarios by farm-type. Interested readers can themselves perform a differential analysis of the farm-types considered.

Table 8. Simulation results by farm-type

Water pricing scenarios	Minimum tillage cereal growers				Conventional farmers with diversified production				Min. tillage and sprinkler irrigation cereal-sugar-beet growers				Direct sowing cereal growers			
	$CIIA_{eco}$	$CIIA_{soc}$	$CIIA_{env}$	CIIA	$CIIA_{eco}$	$CIIA_{soc}$	$CIIA_{env}$	CIIA	$CIIA_{eco}$	$CIIA_{soc}$	$CIIA_{env}$	CIIA	$CIIA_{eco}$	$CIIA_{soc}$	$CIIA_{env}$	CIIA
Baseline scenario	0.18	0.32	0.15	0.66	0.22	0.35	0.17	0.75	0.20	0.32	0.18	0.71	0.15	0.27	0.17	0.59
Health Check	0.14	0.27	0.15	0.56	0.19	0.34	0.17	0.70	0.16	0.29	0.19	0.64	0.13	0.26	0.17	0.57
Pricing instrument: irrigated area																
100 € ha^{-1}	0.11	0.21	0.15	0.48	0.16	0.28	0.17	0.61	0.13	0.23	0.19	0.55	0.10	0.19	0.17	0.47
300 € ha^{-1}	0.06	0.09	0.17	0.32	0.11	0.17	0.18	0.47	0.09	0.10	0.20	0.39	0.06	0.08	0.18	0.32
500 € ha^{-1}	0.04	0.03	0.17	0.24	0.09	0.12	0.18	0.39	0.06	0.04	0.20	0.30	0.03	0.04	0.18	0.25
Pricing instrument: volumetric tariff																
0.01 € m^{-3}	0.12	0.22	0.16	0.50	0.17	0.29	0.17	0.64	0.14	0.23	0.19	0.57	0.12	0.21	0.18	0.50
0.05 € m^{-3}	0.06	0.07	0.17	0.30	0.11	0.16	0.18	0.45	0.08	0.07	0.20	0.35	0.07	0.10	0.18	0.35
0.10 € m^{-3}	0.04	0.03	0.17	0.24	0.09	0.11	0.18	0.38	0.06	0.03	0.20	0.29	0.05	0.05	0.18	0.27
Pricing instrument: two-part tariff system																
50 € ha^{-1} + 0.02 € m^{-3}	0.09	0.13	0.16	0.38	0.14	0.21	0.18	0.53	0.11	0.14	0.20	0.45	0.14	0.21	0.17	0.52
50 € ha^{-1} + 0.06 € m^{-3}	0.05	0.05	0.17	0.27	0.10	0.14	0.18	0.42	0.07	0.05	0.20	0.32	0.12	0.18	0.17	0.47
150 € ha^{-1} + 0.02 € m^{-3}	0.07	0.09	0.17	0.32	0.12	0.17	0.18	0.47	0.09	0.09	0.20	0.38	0.09	0.14	0.18	0.41
150 € ha^{-1} + 0.06 € m^{-3}	0.04	0.03	0.17	0.24	0.09	0.12	0.18	0.39	0.06	0.03	0.20	0.29	0.10	0.12	0.18	0.39
Pricing instrument: block-rate system																
0.01-0.02-0.03 € m^{-3}	0.13	0.19	0.16	0.48	0.16	0.26	0.18	0.59	0.15	0.20	0.20	0.54	0.11	0.15	0.18	0.45
0.04-0.08-0.12 € m^{-3}	0.10	0.07	0.16	0.33	0.12	0.13	0.18	0.43	0.11	0.06	0.20	0.37	0.09	0.09	0.18	0.36

[6] The increased pressure caused by the use of nitrogen and phosphorus is also counterintuitive in terms of the literature. The cause of these results for the case study lies in the relative importance of alfalfa in irrigation, a legume crop with virtually zero balances of nitrogen and phosphorus. Indeed, in most cases, water pricing causes farmers to abandon this irrigated crop and replace it with rain-fed winter cereals, which have a greater demand for chemical fertilizers than alfalfa.

Integrated analysis of public revenue and CIIA

In order to compare the different water pricing systems considered, we have also included public-sector revenue in the analysis. For this purpose, the exercise conducted has been completed by calculating three additional indicators regarding cost-recovery: public revenue per hectare actually irrigated ($PUBR_{irr\_surface}$ measured in € irrigated ha^{-1}), public revenue per cubic meter of water used ($PUBR_{water}$ measured in € m^{-3}) and total public revenue at irrigated area level ($PUBR_{total}$ measured in million €). Table 9 shows the results obtained for these new indicators.

As explained previously, the irrigated district analyzed is fed with surface water managed through publicly owned infrastructures. In such irrigated systems, almost all water delivery costs are fixed (operation and management costs and depreciation costs are not dependant on the amount of water used). This circumstance means the results of $PUBR_{irr\_surface}$ and $PUBR_{water}$ need to be examined with care as cost-recovery indicators, because the overall capacity to generate public revenue depends on the surface area actually irrigated and the amount of water actually used (see the fourth and sixth columns in Table 9). Thus, in order to study the availability of the whole irrigated district to recover the costs of public water services, the only adequate indicator is $PUBR_{total}$.

The results obtained for the new indicators explained above indicate that all the water pricing instruments simulated generate significant increases in public revenues and allow to reduce the quantitiave pressure that agriculture exerts on water bodies (see Table 9, WATER

Table 9. Comparison of public revenue values and CIIA for each water pricing scenario*

Water pricing scenarios	CIIA	$PUBR_{irr\_surface}$ (€ irrigated ha^{-1})	% of the irrigated area devoted to rain-fed crops	$PUBR_{water}$ (€ m^{-3})	WATER (mill. m^{-3})	$PUBR_{total}$ (mill. €)
Baseline scenario	0.68	67.50	0.00	0.0103	248.81	2.55
Health Check	0.61	67.50	9.19	0.0102	227.13	2.31
Pricing instrument: irrigated area						
100 € ha^{-1}	0.52	167.50	23.56	0.0251	192.77	4.84
200 € ha^{-1}	0.43	267.50	39.37	0.0422	145.44	6.13
300 € ha^{-1}	0.37	367.50	51.98	0.0618	107.98	6.67
400 € ha^{-1}	0.32	467.50	60.16	0.0826	85.31	7.04
500 € ha^{-1}	0.28	567.50	65.28	0.1030	72.34	7.45
Pricing instrument: volumetric tariff						
0.01 € m^{-3}	0.54	131.25	19.27	0.0206	194.68	4.00
0.03 € m^{-3}	0.41	224.40	42.56	0.0429	113.64	4.87
0.05 € m^{-3}	0.35	306.00	52.18	0.0642	86.29	5.53
0.07 € m^{-3}	0.32	400.95	57.25	0.0842	77.02	6.48
0.10 € m^{-3}	0.28	551.22	62.42	0.1140	68.76	7.83
Pricing instrument: two-part tariff system						
50 € ha^{-1} + 0.02 € m^{-3}	0.44	234.29	37.28	0.0401	138.55	5.55
50 € ha^{-1} + 0.06 € m^{-3}	0.34	419.81	55.01	0.0833	85.74	7.14
100 € ha^{-1} + 0.02 € m^{-3}	0.40	280.03	44.56	0.0498	117.99	5.87
100 € ha^{-1} + 0.06 € m^{-3}	0.35	472.39	57.90	0.0930	80.92	7.52
150 € ha^{-1} + 0.02 € m^{-3}	0.37	327.05	50.10	0.0597	103.39	6.17
150 € ha^{-1} + 0.06 € m^{-3}	0.30	523.08	60.79	0.1027	75.53	7.75
Pricing instrument: block-rate system						
0.01-0.02-0.03 € m^{-3}	0.52	154.70	21.46	0.0269	171.12	4.60
0.02-0.04-0.06 € m^{-3}	0.45	192.75	35.09	0.0416	113.72	4.73
0.03-0.06-0.09 € m^{-3}	0.40	255.38	42.05	0.0551	101.52	5.59
0.04-0.08-0.12 € m^{-3}	0.36	342.93	45.69	0.0730	96.546	7.05

* $PUBR_{irr\_surface}$: public revenue per hectare actually irrigated; $PUBR_{water}$: public revenue per cubic meter of water used; $PUBR_{total}$: public revenue at irrigated area level.

indicator). However, it should be noted that these enhancements are heterogenous across the water pricing instruments. Differences are mainly linked to farmers' behavior (crop-mix decisions) when facing different water pricing scenarios (see the change in the percentage of irrigated land devoted to rain-fed crops in Table 9). In any case, the most relevant information provided by Table 9 is the comparison of the different water-pricing mechanisms (impact on farm sustainability measures such as the CIIA index) with the rationale of cost-recovery ($PUBR_{total}$ indicator). In this sense, we can see how area fee and block-rate instruments lead to the largest increases in public-sector revenue for any value of the CIIA index. Notwithstanding, the most suitable of these two instruments from a technical point of view is the block-rate system; bearing in mind that the differences in public-sector revenues compared with the area fee are small, but the block-rate system promotes much less water consumption (lower impact on surrounding water ecosystems).

Discussion

Our study shows the practical utility of using composite sustainability indicators as a tool for improving agricultural sector governance. Indeed, the use of composite sustainability indicators makes it possible to address such a complex concept as agricultural sustainability. This entails jointly considering economic, social and environmental indicators. Moreover, public-sector decision-makers may benefit from the use of indices to implement the guidelines derived from the WFD.

Notwithstanding, the main disadvantages of composite indicators also need to be discussed. The two most important problems associated with these indices are (Hansen, 1996; Morse *et al.*, 2001; Ebert and Welsch, 2004; Böhringer and Jochem, 2007): a) the implementation of additive aggregation methods that allow the individual dimensions of sustainability to be compensated for (commensurability), and b) the subjectivity of the weighting process implemented in order to make the assumption that sustainability is a 'social construction' operative. Both circumstances could lead to biased results. Hence, further research is required in order to confirm the accuracy of the approach taken in this study or, on the contrary, to suggest other more precise methods to measure sustainability.

Simulation models based on PMP have also been found useful for *ex-ante* policy analysis. However, taking into account the set of assumptions this modeling technique is based on, the results obtained should be interpreted carefully. Further research in this sense is also required. In line with this, a comparative analysis of results using different matehamatical modeling approaches (PMP, multi-criteria, etc.) are worth suggesting.

Nevertheless, the results suggest the various pricing instruments have different impacts on the socio-economic and environmental indicators and the sustainability index. However, they do have certain effects in common: a) a significant reduction in quantitative pressures (extraction from river flows), and b) generation of damaging economic effects (loss of private irrigation profitability) and social effects (loss of employment generated by the sector). In terms of the composite indicator calculated (CIIA), these dimensional effects would lead to a reduction in the overall sustainability of the agricultural system analyzed here.

Still, it is important to bear in mind that irrigation water pricing is not merely an option, but a mandatory instrument established by the WFD for all member states of the EU. The requirement is based on the premise that implementing cost-recovery pricing will help to achieve the environmental objectives of the Directive (the "good status" of water bodies). However, our results suggest that even though pricing is effective as a means of achieving this objective, it may not be truly efficient, insofar as water pricing policy also generates substantial costs which need to be taken into account, both socioeconomic (profitability and employment) and even environmental (erosion risk, CO_2 capture, etc). This raises some doubts as to the suitability of the lexicographic order of the objectives pursued by the WFD, in which socioeconomic and other environmental objectives are legally subordinated to the improvement of the status of water bodies (Rosenberger *et al.,* 2003; Berbel *et al.,* 2010). In fact, more reseach is required to assess all the impacts of water pricing (both positive and negative) in integrative fashion. Only in this way can we verify whether the implementation of water pricing increases or decreases social welfare.

In spite of the merits of water pricing instruments in general, area fee and block-rate pricing appear to be the most suitable instruments for irrigation water pricing policy, as they produce the smallest reduction in the CIIA index and the highest public-sector revenues. Nevertheless, the most suitable of these two instruments from a technical point of view is the block-rate system; the difference between the two methods in terms of public-sector revenues is small, but the block-rate system promotes less water consumption. This said, it is important to note that excessively high price levels for each block could lead to large-scale abandonment of irrigation due to its lack of socioeconomic sustainability. As indicated previously, this feasible impact should be analyzed together with all the other positive and negative issues in order to achieve a balanced diagnosis that would eventually justify the derogation of the application of water pricing policy as established in the WFD.

Acknowledgments

An early draft of this manuscript was presented at the Second ECORIEGO Workshop, held in Cordoba (Spain) in November, 2010. The authors are indebted to the anonymous reviewers for their useful comments and suggestions, which have led to significant improvements in the paper. The research was co-financed by the Spanish Ministry of Science and Innovation and FEDER through the research project AGRIGOBERSOS (AGL2010-17560-C02-01) and the National Institute for Agricultural and Food Research and Technology (INIA, RTA2008-00022).

References

BAR-SHIRA Z., FINKELSHTAIN I., SIMHON A., 2006. Block-rate versus uniform water pricing in agriculture: an empirical analysis. Am J Agr Econ 88(4), 986-999.

BARTOLINI F., GALLERANI V., RAGGI M., VIAGGI D., 2010. Water management and irrigated agriculture in Italy: multicriteria analysis of alternative policy scenarios. Water Policy 12(1), 135-147.

BERBEL J., RODRÍGUEZ A., 1998. An MCDM approach to production analysis. An application to irrigated farms in Southern Spain. Eur J Oper Res 107(1), 108-118.

BERBEL J., GÓMEZ-LIMÓN J.A., 2000. The impact of water-pricing policy in Spain: An analysis of three irrigated areas. Agric Water Manage 43(2), 219-238.

BERBEL J., VIAGGI D., MANOS B., 2009. Estimating demand for irrigation water in European Mediterranean countries through MCDM models. Water Policy 11(3), 348-361.

BERBEL J., MARTIN-ORTEGA J., MESA, P., 2010. A cost effectiveness analysis of water saving measures for the Water Framework Directive: the case of the Guadalquivir river basin. Water Resour Manag 25, 623-640.

BLANCO M., CORTIGNANI R., SEVERINI S., 2008. Evaluating changes in cropping patterns due to the 2003 CAP reform. Ex-post analysis of different PMP approaches considering new activities. 107th Seminar of the European Association of Agricultural Economists. Seville, Spain.

BÖHRINGER C., JOCHEM P., 2007. Measuring the immeasurable. A survey of sustainability indices. Ecol Econ 63(1), 1-8.

CHD, 2007. Plan hidrológico de la parte española de la demarcación hidrográfica del Duero. Estudio general de la demarcación. CHD, Valladolid. [In Spanish].

CORTIGNANI R., SEVERINI S., 2009. Modeling farm-level adoption of deficit irrigation using Positive Mathematical Programming. Agric Water Manage 96(12), 1785-1791.

DAY R.H., 1963. On aggregating linear programming models of production. J Farm Econ 45(4), 797-813.

EASTER K.W., LIU Y., 2005. Cost recovery and water pricing for irrigation and drainage projects. Agriculture and Rural Development Discussion Paper No. 20. World Bank, Washington, DC.

EBERT U., WELSCH H., 2004. Meaningful environmental indices: A social choice approach. J Env Econ Manag 47(2), 270-283.

FINGER J.M., KREININ M.E., 1979. A measure of 'export similarity' and its possible uses. Econ J 89(356), 905-912.

FOLTZ J.C., MARTIN M.A., LOWENBERG-DEBOER J., 1992. Inclusion of alfalfa (*Medicago sativa* L.) in crop rotations in the Eastern corn belt. Some environmental and economic implications. J Sustain Agr 2(2), 117-133.

FREUDENBERG M., 2003. Composite indicators of country performance: a critical assessment. OECD Science, Technology and Industry Working Papers, 2003/16. OECD, París.

GALLEGO-AYALA J., GÓMEZ-LIMÓN J.A., 2008. Effects of the application of the new CAP and the Water Framework Directive in irrigated agriculture. The case of Arévalo-Madrigal (Ávila) county. ITEA-Inf Tec Econ Agrar 104(3), 335-359. [In Spanish].

GALLEGO-AYALA J., GÓMEZ-LIMÓN J.A., 2009. Analysis of policy instruments for control of nitrate pollution in irrigated agriculture in Castilla y León, Spain. Span J Agric Res 7(1), 24-40.

GIUPPONI C., 2007. Decision support systems for implementing the European Water Framework Directive: The MULINO approach. Environ Modell Softw 22(2), 248-258.

GÓMEZ-LIMÓN J.A., ATANCE I., 2004. Identification of public objectives related to agricultural sector support. J Policy Model 26(8-9), 1045-1071.

GÓMEZ-LIMÓN J.A., RIESGO L., 2004. Irrigation water pricing: differential impacts on irrigated farms. Agr Econ 31(1), 47-66.

GÓMEZ-LIMÓN J.A., GÓMEZ-RAMOS A., 2007. Opinión pública sobre la multifuncionalidad del regadío: el caso de Castilla y León. Econ Agrar Recurs Nat 7(13), 3-25. [In Spanish].

GÓMEZ-LIMÓN J.A., SANCHEZ-FERNANDEZ G., 2010. Empirical evaluation of agricultural sustainability using composite indicators. Ecol Econ 69(5), 1062-1075.

HAJKOWICZ S., 2006. Multi-attributed environmental index construction. Ecol Econ 57(1), 122-139.

HANSEN, J.W., 1996. Is agricultural sustainability a useful concept? Agr Syst 50(1), 117-143.

HAZELL P.B.R., NORTON R.D., 1986. Mathematical programming for economic analysis in agriculture. MacMillan Publ. Co., NY, USA.

HECKELEI T., BRITZ W., 2005. Models based on positive mathematical programming: state of the art and further extensions. In: Modelling agricultural policies: State of the art and new challenges (Arfini F., ed). Ed Monte Università Parma, Parma. pp. 48-74.

HENRY DE FRAHAN B., BUYSSE J., POLOMÉ P., FERNAGUT B., HARMIGNIE O., LAUWERS L., VAN HUYLENBROECK G., VAN MEENSEL J., 2007. Positive mathematical programming for agricultural and environmental policy analysis: review and practice. In: Handbook of operations research in natural resources (Weintraub A., Romero C., Bjørndal T., Epstein R., eds). Springer, NY. pp. 129-154.

HENSELER M., WIRSIG A., HERRMANN S., KRIMLY T., DABBERT S., 2009. Modeling the impact of global change on regional agricultural land use through an activity-based non-linear programming approach. Agric Syst 100(1-3), 31-42.

HOWITT R.E., 1995. Positive mathematical programming. Am J Agr Econ 77(2), 329-342.

HOWITT R.E., 2006. Agricultural and environmental policy models: Calibration, estimation and optimization. Lecture notes, ARE 252. Department of Agricultural Economics, University of California, Davis.

HUETING R., REIJNDERS L., 2004. Broad sustainability contra sustainability: the proper construction of sustainability indicators. Ecol Econ 50(3-4), 249-260.

IGLESIAS E., BLANCO M., 2008. New directions in water resources management. The role of water- pricing policies. Water Resour Res 44, W06417.

INE, 2008. Cuentas satélite del agua. Instituto Nacional de Estadística, Madrid. [In Spanish].

JOHANSSON R.C., TSUR Y., ROE T.L., DOUKKALI R., DINAR A., 2002. Pricing irrigation water: a review of theory and practice. Water Policy 4(2), 173-199.

MOLLE F., BERKOFF J., 2007. Water pricing in irrigation: mapping the debate in the light of experience. In: Irrigation water pricing. The gap between theory and practice (Molle F., Berkoff J., eds). Ed CABI, Wallingford. pp. 21-93.

MORSE S., MCNAMARA N., ACHOLO M., OKWOLI B., 2001. Sustainability indicators: the problem of integration. Sustain Dev 9(1), 1-15.

MUNDA G., 2005. Measuring sustainability: a multi-criterion framework. Env Develop Sustain 7(1), 117-134.

OECD-JRC, 2008. Handbook on constructing composite indicators. Methodology and user guide. OECD, Paris.

QIU H.J., ZHU W.B., WANG H.B., CHENG X., 2007. Analysis and design of agricultural sustainability indicator systems. Agr Sci China 6(4), 475-486.

RANDALL A., 1981. Property entitlements and pricing policies for a maturing water economy. Aust J Agr Econ 25(3), 195-220.

RIGBY D., WOODHOUSE P., YOUNG T., BURTON M., 2001. Constructing a farm-level indicator of sustainable agricultural practice. Ecol Econ 39(3), 463-478.

RÖHM O., DABBERT S., 2003. Integrating agri-environmental programs into regional production models: an extension of positive mathematical programming. Am J Agr Econ 85(1), 254-265.

ROSENBERGER R.S., PETERSON G.L., CLARKE A., BROWN T.C., 2003. Measuring dispositions for lexicographic preferences of environmental goods: integrating economics, psychology and ethics. Ecol Eco 44(1), 63-76.

SANDS G.R., PODMORE T.H., 2000. A generalized environmental sustainability index for agricultural systems. Agr Ecosys Envir 79(1), 29-41.

TSUR Y., DINAR A., 1997. The relative efficiency and implementation costs of alternative methods for pricing irrigation water. World Bank Econ Rev 11(2), 243-262.

VAN CALKER K.J., BERENTSEN P.B.M., ROMERO C., GIESEN G.W.J., HUIRNE R.B.M., 2006. Development and application of a multi-attribute sustainability function for Dutch dairy farming systems. Ecol Econ 57(3), 640-658.

VAN CAUWENBERGH N., BIALA K., BIELDERS C., BROUCKAERT V., FRANCHOIS L., GARCIA CIDAD V., HERMY M., MATHIJS E., MUYS B., REIJNDERS J. *et al.*, 2007. SAFE - a hierarchical framework for assessing the sustainability of agricultural systems. Agr Ecosys Envir 120(2-4), 229-242.

VARELA-ORTEGA C., SUMPSI J.M., GARRIDO A., BLANCO M., IGLESIAS E., 1998. Water pricing policies, public decision-making and farmers' responses: Implications for water policy. Agric Econ 19(2), 193-202.