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THE ROLE OF GROUNDWATER-DEPENDENT ECOSYSTEMS IN GROUNDWATER MANAGEMENT

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ABSTRACT. Ecosystems provide a wide range of services essential for a proper environmental, economic, and social performance. While the estimated global value of ecosystem services in 2014 is very significant, the annual loss of ecosystem services value is alarming. Our paper focuses on groundwater-dependent ecosystems (GDEs), some very important to society, which are under threat due to groundwater overexploitation. Considering the ecosystem health/status function is essential for sound groundwater regulation policy. The paper assesses the conjunctive management of groundwater and GDEs both in theory and in a relevant case study, using a certain type of an ecosystem health function. The theoretical results demonstrate how the change in the slope values of a general ecosystem health function affects the optimal groundwater management policy. The analysis also suggests a change in groundwater management strategies as a function of the value of the ecosystem. The theoretical findings are corroborated with data from an aquifer in Spain and its associated GDE—the Tablas de Daimiel Wetland. The paper highlights theoretically and empirically the necessity for a better understanding of GDEs behavior. It calls for groundwater regulation to protect these resources.

KEY WORDS: Optimal control, groundwater-dependent ecosystems (GDEs), groundwater management, ecosystem health function.

1. Introduction. The global value of ecosystem services was estimated in 2014 to range between 125 and 145 trillion dollars per year. At the same time loss of ecoservices in the past 15 years was estimated at 4.3–20.2 trillion dollars per year (Costanza et al. [2014] following the first estimate by Costanza et al. [1997]). Estimated flow value per hectare per year of ecosystem services for water-related ecosystems ranges between \$4267 and \$140,147 respectively, for lakes and rivers, and for wetlands (Costanza et al. [2014]). These values declined from previous estimates (in Costanza et al. [1997]) due to changes in land use and, in the case of water-related ecosystems, changes in water extractions.

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The societal value of groundwater resources has been better understood recently due to increasing evidence of the variety of services it provides. Moreover, groundwater is the most important source of fresh water worldwide with around 30% of the fresh water stored in aquifers (Howard and Merrifield [2010]). Groundwater reservoirs serve as a buffer for supply under extremely scarce and variable conditions (Tsur and Graham-Tomasi [1991]) because they are used for storage in periods of plenty and release in periods of scarcity. Additionally, one of the most important roles of groundwater is its support for ecosystems that serve as crucial links between nature and society.

Recent economic literature has analyzed groundwater regulation for achieving efficient management of this resource (Gisser and Sánchez [1980], Dixon [1989], Provencher [1993], Brill and Burness [1994], Knapp and Olson [1995], Burness and Brill [2001]). Most of these studies only address the congestive externality from the open-access nature of groundwater. Part of the literature analyzes groundwater management when a negative externality exists in the form of groundwater quality deterioration due to large extractions (Tsur and Graham-Tomasi [1991], Dinar [1994], Dinar [1997], Yadav [1997], Dinar and Xepapadeas [1998], Koundouri [2000], Roseta-Palma [2002], Shah et al. [2000], or Konikow and Kendy [2005]). Other studies focus on the existence of additional environmental costs (Esteban and Albiac [2011]) or additional costs related to interactions between groundwater and surface water (Young et al. [1986], Kuwayama and Brozovic [2013], or Roumasset and Wada [2013]).

However, the impacts of groundwater overexploitation on the health of groundwater-dependent ecosystems (GDEs), which represent another relevant externality, are still scantly analyzed. Some examples of analyzing GDEs can be found in Brown et al. [2011], KlØve et al. [2011a, b], Howard and Merrifield [2010], van der Kamp and Hayashi [2008], Murray et al. [2008], Eamus and Froend [2006], or Stromberg et al. [1996]. Most of these studies analyze the role of GDEs, their interaction with groundwater, and the importance of their preservation. But none of these studies has properly incorporated the economic role of ecosystems, with their associated ecological services, as an essential element in the groundwater management strategy. Moreover, previous literature did not address the important issue of the link between the shape of the health, or status,[1] function of the ecosystem and the optimal policy needed for regulation ecosystem sustainability.

GDEs are ecosystems that rely mainly on water stored in aquifers. The water is essential for maintaining their ecological processes and their status (Government of South Australia [2010]). A large number of GDEs can be identified and classified as (Sinclair [2001], Foster et al. [2006]): (1) terrestrial vegetation and fauna that have seasonal dependence on groundwater; (2) ecosystems dependent on river streams that are eventually fed by groundwater; (3) aquifer and cave ecosystems; (4) wetlands and lakes that are mainly fed by aquifers' water; and (5) estuarine and coastal lagoons that depend on the discharge of groundwater floods. All these ecological systems provide various ecosystem services to society (Bergkamp and Cross [2006]) and the provision level depends on the groundwater characteristics such as water quality, aquifer level, and/or water pressure (Sinclair [2001]).

The change in environmental conditions largely affects the ecosystems. Several works (Cropper [1976], Tsur and Zemel [1995], Gollier et al. [2000], Peterson et al. [2003], Créporin [2007], Brozovic and Schlenker [2011], Polasky et al. [2011], Crépin et al. [2012], Lade et al. [2013]) have addressed various aspects of ecosystem behavior under situations of regime shift and catastrophic events with uncertain ecosystem thresholds and irreversibility. When conditions approach a critical level the status of the ecosystems is altered. This modification is not necessarily gradual, and abrupt switches can occur (Scheffer et al. [2001]). Therefore, an important question is how gradual is the ecosystem's response to deterioration in the habitat conditions. Despite of the extended conception that ecosystems behavior does not follow linear approaches (Chapin III et al. [2000], Burkett et al. [2005], Barbier et al. [2008]), continuous linear specifications are still considered by several authors as a good approach to analyze changes in ecosystems status (Schwartz et al. [2000], Poff et al. [2010], Williams et al. [2011]). Recently, some authors have also argued that there is no convincing evidence in the literature that supports the existence of nonlinear relationships in ecosystems neither the existence of stark changes in ecosystems (Mac Nally et al. [2014], Capon et al. [2015]).[2] As Burkett et al. [2005:386] point out "*A better understanding of linear and nonlinear ecosystem processes and patterns will improve science-based management of natural resources.*"

In the case of groundwater, the depletion of the water level generates significant impacts on GDEs. Ecosystems' responses can be different, depending on the ecosystem type (which is reflected in its health function). The results of this paper demonstrate that under a linear and continuous ecosystem health function, groundwater regulation may or may not be necessary, depending on the function's parameters. This result clearly calls for a better understanding and need for more information about the ecosystems' behavior, which is essential for designing a management policy to pursue the protection of ecosystems and the production of their ecological goods and services (Boyd [2011]).

This paper contributes to the existing literature by introducing the GDEs as an additional groundwater user, characterized by a behavioral or health function, which is an integrated part of an optimal groundwater management model ("social planner regime"). The social planner internalizes two externalities: (1) extraction/congestion externality (the damage that one user imposes on the other users); and (2) environmental externality (the benefits to the society from GDEs services). The environmental externality depends on the type of GDE, which is defined by the ecosystem health function. The impact of groundwater reduction on the ecosystems generates different ecological status levels depending on the response of the ecosystems to habitat changes. In this paper, we focus on the well-known case of a linear and continuous specification of ecosystem behavior to changes in their habitat conditions. This function, largely analyzed and supported with empirical evidence in the literature (Cuffney et al. [2010], O'Farrell et al. [2011]) allows us to develop a theoretical model and also a numerical application for a real case study. Our main contribution is the analysis of how a linear and continuous specification of the health function can, in fact, alter the optimal groundwater management regime and its impacts on social welfare and the ecosystem status.

We show how the "social planner regime" solution (optimal water table level and rate of extraction) is influenced by the response of the ecosystem to habitat modifications and the resulting economic value. The paper illustrates, both theoretically and empirically, that the slope of the ecosystem status function as well as the economic value of the ecosystem affect the optimal social planner solution. These results highlight the importance of the ecological and social consequences of changes in biodiversity and how managers should internalize these impacts into water management. As could be expected the results highlight the necessity for regulation when positive environmental externalities are taken into account. This result is also well supported in the literature, however the main difference is that in most of the previous studies the environmental externality is related to groundwater quality or additional groundwater costs, based on damages to the ecosystem.

The theoretical model is empirically validated in one of the most important aquifers in Spain—the Western la Mancha aquifer and its connected GDE—Tablas de Daimiel wetland. The increasing extraction rate in the region during the past 52 years has driven the aquifer to a severe level of overexploitation. This depletion created serious damages to the wetland, which depends on the aquifer; we quantify such losses. The wetland provides buffer services to the aquifer and also economic services (e.g., recreation) to the society in surrounding communities.

The paper proceeds as follows: Section 2 reviews the main ecosystem responses to deterioration in environmental conditions. In Section 3 we set up the groundwater model in which the positive environmental externality of GDEs is included and optimal management conditions are derived. We test the theoretical findings in the Western la Mancha aquifer and its Tablas de Daimiel wetland in Section 4. Discussion and conclusions are provided in Section 5.

2. Ecosystem health functions. External conditions generate significant changes in the health of GDEs. GDEs respond differently to changes in their natural habitat (Vitousek et al. [1997], Sheffer et al. [2001], Tilman et al. [2001]). One important external condition that can largely affect the health of several GDEs is the reduction in groundwater levels (Scheffer et al. [2001]). Three main status functions are suggested in the literature to represent shifts in the ecosystem health due to alterations in their natural habitat. Depending on the ecosystem, the response to changes in external conditions could be:[3]

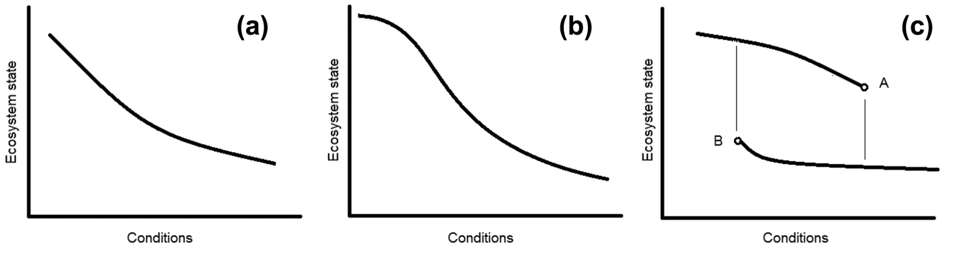

FIGURE 1. Shifts in various ecosystem states depending on the ecosystem type.

Source: Adapted with minor modifications from Scheffer et al. [2001].
Note: Conditions worsen from left to right.

(1) **Smooth and continuous shift**: alterations in the external conditions generate a decrease in the ecosystem state (e.g., population) but the ecosystem will not totally disappear. Figure 1a shows the graphical representation of this type of behavior. This is a typical representation of a riparian ecosystem.
(2) **Nonlinear declining shift**: ecosystems are inert over certain range of external conditions but as long as a critical level is reached a stronger ecosystem response takes place (Figure 1b). An example of such behavior is a forest; when conditions get extreme, most of the trees die but some vegetation, such as bushes and other foliage remains.
(3) **"Folded" backward shift**: ecosystems respond to deterioration in external conditions following a curve that is "folded" backward. The ecosystem has two stable states, depending on the environmental conditions (Figure 1c). If the ecosystem is on the upper line and close to point "A," minor changes in the conditions may induce a catastrophic switch to the lower line. To switch again to the upper line, the conditions need to be reversed far enough to reach point "B." An example of this representation is a deserted area in which vegetation cover is lost; water and winds erode the soil making the area hostile to vegetation and much more hostile for recolonizing seedlings.

The behavior of almost all known ecosystems fits one of these three representations. In this paper we follow a smooth and continuous linear function to represent the ecosystem behavior.

2.1. Mathematical representation. One of the challenges of this paper is to have our theoretical model and results supported by an empirical application with data from an aquifer and its related ecosystem. Although many ecologists defend the third function (Folded Backward Shift), implying some degree of irreversibility in ecosystem collapse for extreme conditions, we will follow a smooth and continuous linear function given the information challenges described in the next section. With

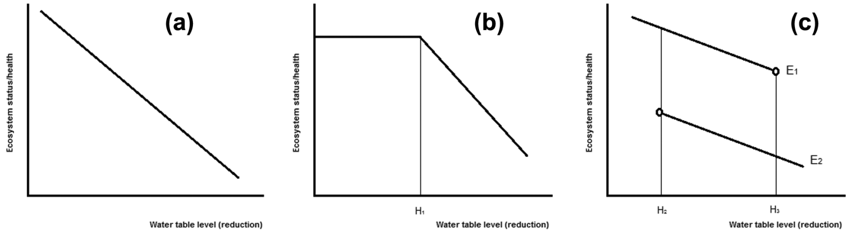

FIGURE 2. Linear ecosystem health functions.

the scantly data available we have been able to plot a linear relationship between the flooded area of a wetland and the decrease in the aquifer storage. We do not have evidence (long enough time horizon or temporal data resolution) to empirically report the cases of discontinuous functions (case 1b or 1c), and neither information to state the possible thresholds of those functions.

Following the previous discussion, three linearized mathematical functions are defined and described (Figure 2), although we will refer in our analytical and empirical model only to the smooth and continuous function.

The first ecosystem representation (Figure 2a) corresponds to Figure 1a above. The health of the ecosystem due to a reduction in the water table level is represented by a negatively sloped linear function. This function suggests that as much as the water table decreases the ecosystem health also deteriorates.[4]

$$D_1\left(H\left(t\right),\ t\right) = \left[\sigma_1 - \rho_1\left(S_L - H\left(t\right)\right)\right], \tag{1}$$

where $D_1(H(t),\ t)$ is the ecosystem population (or ecosystem health), which depends on the water table level in period t; σ_1 and ρ_1 are the ecosystem status function parameters (where $\sigma_1 > 0$ and $\rho_1 > 0$); S_L is the elevation of the surface; and $H(t)$ is the water table level at time t. As external conditions worsen (in our case, as water level in the aquifer drops) the difference between the surface elevation and the water table $(S_L - H(t))$ increases and the ecosystem health $D_1(H(t),\ t)$ decreases. We use this representation in our empirical specification.

The second ecosystem representation (Figure 2b) corresponds to Figure 1b above (step-wise declining function). When the water table level is above a critical threshold (H_1) the ecosystem is not affected. However, once this critical level is reached the ecosystem health decreases linearly at a constant rate per unit of water table reduction. The mathematical representation of this behavior is

$$D_2\left(H\left(t\right),\ t\right) = \begin{cases} \sigma_2 & \text{if } \ H\left(t\right) \geq H_1 \\ \sigma_2 - \rho_2\left(S_L - H\left(t\right)\right) & \text{if } \ H\left(t\right) < H_1 \end{cases} \tag{2}$$

with $D_2(H(t),\ t)$ representing the ecosystem health level under a step declining function; σ_2 and ρ_2 are the ecosystem status function parameters (with $\sigma_2 > 0$ and $\rho_2 > 0$); and H_1 is the water table's critical threshold.

The third representation is the "backward folded" function (Figure 2c), which corresponds to Figure 1c above. This function is broken into two branches, or two different functions (E_1 and E_2), where two thresholds can be identified (H_2 and H_3). When the ecosystem is on the upper part (upper function, E_1) and the water table decreases, reaching the critical level H_3, the ecosystem switches to the lower part (lower function E_2). This switch causes significant reduction in the ecosystem health and may even lead to the extinction of some of the ecosystem species. The ecosystem will stay at this lower function (E_2) until the conditions significantly improve. If the water table level increases, reaching the threshold H_2, then the opposite process will take place. The ecosystem health will then recover moving again to the upper part. The mathematical representation of this behavior is

$$(3)\ D_3\left(H\left(t\right),t\right) = \begin{cases} \begin{cases} \sigma_3 - \rho_3(s_L - H(t)) & \text{if} \quad H(t) \geq H_2 \\ \sigma_3 - \rho_3(s_L - H(t)) & \text{if} \quad H_2 < H(t) < H_3 \text{ and } \ E_t \geq \overline{E} \end{cases} \\ \begin{cases} \sigma_4 - \rho_4(s_L - H(t)) & \text{if} \quad H_2 < H(t) < H_3 \text{ and } \ E_t < \overline{E} \\ \sigma_4 - \rho_4(s_L - H(t)) & \text{if} \quad H(t) \leq H_3 \end{cases} \end{cases},$$

where $D_3(H(t),\ t)$ is the ecosystem health under a "folded" backward function. The terms σ_j and ρ_j (with $j = 3,\ 4$) are the function parameters ($\sigma_j > 0$ and $\rho_j > 0$). The term H_2 is the lower function (line E_2) critical threshold, and H_3 is the upper function (line E_1) critical threshold. The term $\overline{E}$ represents a critical ecosystem health level, which splits the ecosystem health between a higher branch (E_1) and a lower branch (E_2).

3. Optimal groundwater model integrating an ecosystem response function. We model the optimal management of an aquifer linked with a GDE as a social planner problem. The main objective of the social planner is to maximize the social benefits from extracting groundwater. The social benefits depend on the benefits to private users and the social benefits from GDEs, which require certain level of groundwater to guarantee their survival and allow the provision of measurable amount of goods and services. The social planner, thus, pursues to protect GDEs, which are being seriously affected by otherwise excessive groundwater extraction.

A dynamic optimal control model linking economic, hydrologic, agronomic, and environmental variables is developed to characterize the groundwater system's behavior. Following the framework proposed by Gisser and Sánchez [1980], we incorporate a GDEs externality to the model. We also extend the work by Esteban and Albiac [2011] in three aspects. First, Esteban and Albiac [2011] use a scalar to represent the value of the GDE in the objective function while we use an explicit empirical function; second, the ecosystem health function has been estimated by using the available data for the Tablas de Daimiel wetland (we plot a linear decreasing relationship between the wetland flood area and the water table level); and third, Esteban and Albiac [2011] do not use a water demand function by the agricultural sector, while we estimate and introduce such function into our model.

3.1. Model components. The groundwater demand function (W) depends on the water price (P) and represents the value of the marginal physical product of irrigation water

$$W = g + k \cdot P, \tag{4}$$

where g and k are the demand function parameters (with $g > 0$ and $k < 0$).

The supply function represents the unit cost of the groundwater extraction

$$\bar{P} = C_0' + C_1' \cdot (S_L - H)\,, \tag{5}$$

where C_0' represents the fixed costs due to the hydrological cone, and C_1' is the marginal pumping cost. The term S_L is the elevation of the surface above sea level, and H is the water table elevation above sea level. The previous equation can be simplified to

$$\bar{P} = C_0 + C_1 \cdot H, \tag{6}$$

where $C_0 = C_0' + C_1' \cdot S_L$ and $C_1 = -C_1'$.

The hydrological characteristics of the aquifer relate the water table with time. This function depends on the aquifer recharge and the level of the aquifer

$$AS \cdot \dot{H} = R + (\alpha - 1) \cdot W, \tag{7}$$

where R is natural recharge, 'α' is the return flow coefficient of the applied irrigation water, and AS is the area of the aquifer (A) multiplied by the storativity coefficient (S).[5] The term AS represents the water availability flow into the aquifer.

The environmental externality relates the health of the ecosystem with the depletion of the aquifer (measured as the difference between the surface level and the water table level). This externality is a positive one, representing the benefits that society obtains from the goods and services provided by ecosystem[6]

$$D\,(H) = \sigma - \rho \cdot (S_L - H)\,, \tag{8}$$

where $D(H)$ is the ecosystem status function representing a smooth and continuous linear specification (defined in Section 2).

3.2. The optimization problem and optimality conditions. The farmers' optimization problem consists of a maximization of the present value of their future income stream; this is the private optimal control regime. Farmers do not internalize the benefits provided by the ecosystems (GDEs) and maximize their private benefits from groundwater consumption minus the pumping costs. Under the private problem farmers take into account the negative impact that their groundwater extraction imposes on other users (extraction externality)[7]

$$\begin{aligned} &\text{Max } \Pi = \int_0^\infty e^{-rt} \cdot \left[\frac{1}{2k} \cdot W^2 - \frac{g}{k} \cdot W - (C_0 + C_1 \cdot H) \cdot W\right] dt, \\ &\text{s.t. } \dot{H} = \frac{[R + (\alpha - 1)\, W]}{AS}, \\ &H\,(0) = H_0, \end{aligned} \tag{9}$$

where Π is the farmers' total private profit. The farmers' total revenue $(\frac{1}{2k} \cdot W^2 - \frac{g}{k} \cdot W)$ is the area under the inverse irrigation water demand curve (see equation (4)). And $(C_0 + C_1 \cdot H) \cdot W$ is the total cost of groundwater pumping (see equation (6)).

However, the objective of the social planner is to maximize the social benefits and not just the private ones. The social benefit stream is the private net income plus the GDEs externality representing the benefits from the ecosystems goods and services. The social planner optimal control problem is then

$$\begin{aligned} &\text{Max } SP = \int_0^\infty e^{-rt} \cdot \left[\frac{1}{2k} \cdot W^2 - \frac{g}{k} \cdot W - (C_0 + C_1 \cdot H) \cdot W \right. \\ &\qquad\qquad \left. + \, \xi \cdot (\sigma - \rho \cdot (S_L - H))\right] dt, \\ &\text{s.t. } \dot{H} = \frac{[R + (\alpha - 1)\, W]}{AS}, \\ &H\,(0) = H_0, \end{aligned} \tag{10}$$

where SP represents the social planner problem. The environmental externality is $\xi \cdot (\sigma - \rho \cdot (S_L - H))$, with ξ representing the economic value per unit of ecosystem status.

The Hamiltonian of the social planner problem is expressed as

$$H = -e^{(-rt)} \left[\frac{1}{2k} \cdot W^2 - \frac{g}{k} \cdot W - (C_0 + C_1 \cdot H) \cdot W + \xi \cdot (\sigma - \rho \cdot (S_L - H)) \right]$$

$$+ \lambda \cdot \left[\frac{(R + (\alpha - 1) \cdot W}{AS} \right]. \tag{11}$$

First-order conditions (FOC) of the Hamiltonian are

$$\frac{\partial \mathcal{H}}{\partial W} = -e^{-rt} \left[\frac{1}{k} \cdot W - \frac{g}{k} - (C_0 + C_1 \cdot H) \right] + \lambda \cdot \left(\frac{(\alpha - 1)}{AS} \right) = 0, \tag{12}$$

$$\frac{\partial \mathcal{H}}{\partial H} = \dot{\lambda} = - \left[-e^{-rt} \left(-C_1 \cdot W + \xi \cdot \rho \right) \right], \tag{13}$$

$$\frac{\partial \mathcal{H}}{\partial \lambda} = \dot{H} = \frac{[R + (\alpha - 1) \cdot W]}{AS}, \tag{14}$$

$$\lim_{t \to \infty} \lambda (t) \cdot H (t) = 0. \tag{15}$$

Following the methodology of Gisser and Sánchez [1980] from the FOC (equation (12)) we can obtain the value of

$$\lambda = \frac{AS}{(\alpha - 1)} \left[e^{-rt} \cdot \left(\frac{1}{k} \cdot W - \frac{g}{k} - (C_0 + C_1 \cdot H \right) \right]. \tag{16}$$

Equation (16) is differentiated with respect to time (t), and then we can replace $\dot{\lambda}$ by the expression from equation (13). Rearranging terms yields the following equation

$$\frac{1}{k} \cdot \dot{W} - C_1 \cdot \dot{H} = \frac{(\alpha - 1)}{AS} \cdot (-C_1 \cdot W + \xi \cdot \rho) + \frac{1}{k} \cdot r \cdot W$$

$$- \frac{g}{k} \cdot r - C_0 \cdot r - C_1 \cdot H \cdot r. \tag{17}$$

Replacing $\dot{H}$ with the expression from equation (14) and rearranging terms yields

$$\dot{W} = \frac{R}{AS} \cdot k \cdot C_1 - \frac{(\alpha - 1)}{AS} \cdot k \cdot \xi \cdot \rho$$

$$+ r \cdot W - g \cdot r - C_0 \cdot k \cdot r - C_1 \cdot k \cdot r \cdot H. \tag{18}$$

Equations (14) and (18) allow us to establish a system of two differential equations with two unknowns

$$\dot{W} = r \cdot W - n \cdot H + N + N' \tag{19}$$

and

$$\dot{H} = m \cdot W + M \tag{20}$$

with $n = C_1 \cdot k \cdot r$, $N = \frac{k \cdot C_1 \cdot R}{AS} - r \cdot g - C_0 \cdot k \cdot r$, $N' = \frac{(\alpha - 1) \cdot k}{AS} \cdot \xi \cdot \rho, m = \frac{(\alpha - 1)}{AS}$, and $M = \frac{R}{AS}$.

Differentiating again equation (19) with respect to time yields

$$\ddot{W} = r \cdot \dot{W} - n \cdot \dot{H}. \tag{21}$$

Substituting equation (20) into equation (21) yields

$$\ddot{W} - r \cdot \dot{W} - n \cdot m \cdot W = 0. \tag{22}$$

The solution to this differential equation is

$$W(t) = A \cdot e^{tx_1} + B \cdot e^{tx_2}. \tag{23}$$

The parameters A and B are equation constants,[8] and x_1, x_2 are the roots of the polynomial equation $(x^2 - r \cdot x - m \cdot n = 0)$. Integrating equation (20) we obtain the following expression

$$H(t) = \frac{m \cdot A}{x_1} \cdot e^{tx_1} + \frac{m \cdot B}{x_2} \cdot e^{tx_2}. \tag{24}$$

Setting $\dot{W} = 0$ and $\dot{H} = 0$ in equations (19) and (20) provides the solutions of the optimal extractions and water table level under the social planner problem

$$W(t) = A \cdot e^{tx_1} + B \cdot e^{tx_2} - {}^{M}/_{m}, \tag{25}$$

$$H(t) = \frac{m \cdot A}{x_1} e^{tx_1} + \frac{m \cdot B}{x_2} e^{tx_2} + \frac{N + N' - r \cdot (M/m)}{n}. \tag{26}$$

With some simplifying assumptions[9] the water extractions and the water table level expressions in the social planner optimal control problem are, respectively,

$$W(t) = -\frac{R}{(\alpha - 1)} + \left\{ \frac{R}{(\alpha - 1)} + g + k \cdot C_0 + k \cdot C_1 \cdot H_0 - \frac{(\alpha - 1) \cdot k \cdot \xi \cdot \rho}{r \cdot AS} - \frac{R \cdot k \cdot C_1}{r \cdot AS} \right\} \cdot e^{tx_2}, \tag{27}$$

$$H(t) = \frac{R}{r \cdot AS} + \frac{-(\alpha - 1) \cdot (g + k \cdot C_0) - R}{(\alpha - 1) \cdot k \cdot C_1} + \frac{(\alpha - 1) \cdot \xi \cdot \rho}{r \cdot AS \cdot C_1} + \left\{ \frac{R + (\alpha - 1) \cdot (g + k \cdot C_0) + (\alpha - 1) \cdot k \cdot C_1 \cdot H_0}{(\alpha - 1) \cdot k \cdot C_1} - \frac{(\alpha - 1) \cdot \xi \cdot \rho}{r \cdot AS \cdot C_1} - \frac{R}{r \cdot AS} \right\} \cdot e^{tx_2}, \tag{28}$$

where x_2 is

$$x_2 = \frac{r - (r^2 - 4 \cdot m \cdot n)^{1/2}}{2} = \frac{r - \sqrt{r^2 - 4 \cdot \frac{(\alpha - 1)}{AS} \cdot C_1 \cdot k \cdot r}}{2}. \tag{29}$$

This expression is similar to the one obtained by Gisser and Sánchez [1980].[10] The root of the exponent, from the differential equation (equation (23)), does not change when environmental externalities are considered.

Gisser and Sánchez [1980] state that in the case of a large aquifer, when the values of AS are high enough, the term $\frac{R \cdot k \cdot C_1}{r \cdot AS} \cdot e^{tx_2}$ in equation (27), and the terms $\frac{R}{r \cdot AS}$ and $\frac{R}{r \cdot AS} \cdot e^{tx_2}$ in equation (28), can be eliminated because the denominator will be large enough to make them close to zero. Due to this simplification, Gisser and Sánchez concluded that regulation is not justified in groundwater management. Their empirical findings confirm how the optimal results of the water table and extractions under no regulation ("free-market or farmers' private problem") and under regulation ("optimal control") are quite similar. Therefore, regulation is neither necessary nor efficient according to Gisser and Sánchez [1980].

However, in the social planner problem, when GDEs are also groundwater users, the expressions of the optimal water table and optimal extractions have additional terms that depend on the ecosystems' health. In the case of the optimal extractions (equation (27)), a new term $\frac{(\alpha - 1) \cdot k \cdot \xi \cdot \rho}{r \cdot AS} \cdot e^{tx_2}$ is introduced. In the case of the optimal water table level (equation (28)), two additional new terms are introduced, $\frac{(\alpha - 1) \cdot \xi \cdot \rho}{r \cdot AS \cdot C_1}$ and $\frac{(\alpha - 1) \cdot \xi \cdot \rho}{r \cdot AS \cdot C_1} e^{tx_2}$.

Under the social planner problem, the optimal levels of the water table and the extractions depend on the economic value of the ecosystem (ξ). The higher the economic value the higher the impact on the optimal solution. But the optimal solution also depends on the ecosystem's health function slope (ρ). This means that the greater the absolute value of the slope the higher the impact on the ecosystem's health due to groundwater depletion. It is also important to take into account, that under the linear assumptions of this model the optimal paths for the water table and the level of extractions do not depend on the ecosystem function intercept (σ).

Finally, observing the equation of the water table level (equation (28)) it is possible to identify the expression $\frac{(\alpha-1)\cdot\xi\cdot\rho}{r\cdot AS\cdot C_1}$, which has a positive sign. It is important to realize that this expression is not time-related. This means that when GDEs are accounted for in the model, the water table level should reach initial higher values compared with cited earlier the results when no environmental externalities are internalized.

Other studies have also demonstrated that the economic value of the ecosystems (or the economic value of the damage to an ecosystem)[11] could modify the optimal groundwater management. However, our results differ from these studies because we clearly show how ecosystem behavior function affects the optimal management. Using a continuous linear health function we have demonstrated that the internalization of ecosystems in groundwater management affects the optimal paths for both the extractions and the water table level.

At this point we can introduce two propositions:

Proposition 1. *The economic value of a GDE directly affects the management of the aquifer water in the optimal solution in terms of both the extractions and the water table level over time. The higher the GDE value, the lower the optimal extractions and the higher the water table optimal level. (See Appendix A for proof).*

Proposition 2. *The rate of impact of external conditions on the status of the GDE directly affects the management of the aquifer water in the optimal solution, in terms of both the extraction and the water table levels over time. The higher the GDE sensitivity, the lower the optimal extractions and the higher the water table optimal level. (See Appendix A for proof).*

Following Proposition 1, we can conclude that a policy intervention in groundwater management is justified and necessary when GDEs are highly valuable. In addition, Proposition 2 also suggests that not just the economic value of the GDEs but also the health function (represented by the ecosystems' function slope) is a key element to take into account in order to achieve an efficient groundwater management and protect GDEs.

These two propositions suggest some immediate policy implications. The fact that the ecosystem function slope appears in the optimal solutions in both water extractions and water table level highlights the need for better knowledge about the GDEs response function. Depending on the differences in the ecosystem's behavior (namely the slope of the ecosystem's health function), the optimal groundwater management and the optimal policy can be substantially different. We will demonstrate these points using the case of the Tablas de Daimiel wetland in Western la Mancha (WLM).

4. Empirical application. The theoretical model is empirically tested in the Western la Mancha (WLM) aquifer, located in southeast Spain. The significant increase in the water extractions due to the development of intensive irrigated agriculture in the region has led to several negative impacts on this aquifer and also on several GDEs in the region. The decrease in the water table affects the Tablas de Daimiel wetland. The health of the wetland is linked to the aquifer's water level, and a drastic depletion of aquifer water level significantly decreases the wetland flood area, which is the measure of its ability to provide ecosystem services.

4.1. Western la Mancha aquifer and Tablas de Daimiel wetland. The WLM aquifer is one of the largest groundwater bodies in Spain. The aquifer has an area of nearly 5500 km^2 with estimated water storage of 12,000 million cubic meters (Mm^3) (Martínez-Santos et al. [2008]). Climatic conditions are continental and semi-arid, characterized by hot temperatures during long summer periods and warm temperatures in the short winter periods. The long-term mean annual precipitation is around 415 mm/year with seasonal precipitation, and long dry periods.

The WLM aquifer supplies around 90% of the irrigation water used in the region. The development of intensive irrigated agriculture during the 1970s has caused a depletion of 3000 Mm^3 in the aquifer storage. This means an accumulated decrease of 24 m in the water table level. The irrigation acreage has expanded from 30,000 ha in 1960s to nearly 200,000 ha in 2000s (Esteban and Albiac [2011], Esteban and Albiac [2012]). Due to this large depletion in the aquifer level, the WLM was officially declared overexploited in 1994, including several problems of water quality that were also identified (IGME [2004]). Furthermore, one of the most important impacts of this overexploitation is the damage caused to several ecosystems surrounding WLM, including the Tablas de Daimiel wetland.

The Tablas de Daimiel wetland is a marshy area covering 2000 ha over the WLM aquifer. The wetland was declared a National Park in 1973, UNESCO biosphere reserve in 1981, part of the RAMSAR convention in 1982, and Special bird and Natura 2000 protected area by the European Union regulation. Tablas de it Daimiel has very rich aquatic ecosystems with unique species of flora and fauna and is also a place for migrant waterfowl between Europe and Africa. Since the 1980s, Tablas de Daimiel has decreased its annual flood acreage (Figure 3). Even though the wetland areas are highly dependent on the rainfall (see Figure C2) the aquifer depletion is strongly correlated with the wetland flood area.

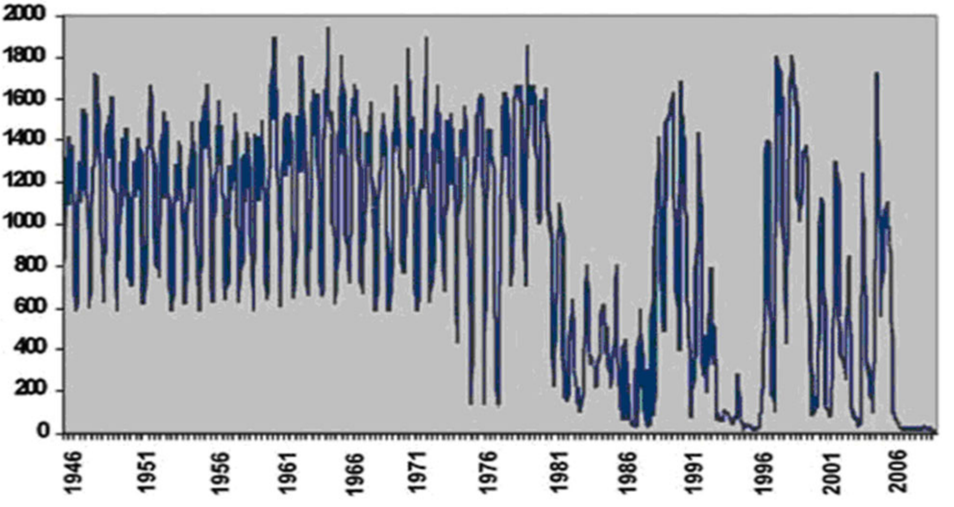

FIGURE 3. Historical evolution of the flood area in Tablas de Daimiel (ha).

Source: Ruíz de la Hermosa [2011].

4.2. Data. The parameters used in the empirical application are depicted in Table 1. The hydrological data are from Martínez-Santos et al. [2008] and IGME [2004].

The existing literature presents different estimations for the WLM groundwater demand function (see Appendix B). The water demand function in WLM has been estimated using data from PEAG [2008], approximating a linear relationship: $k = -0.7272$ and $g = 726.71$ (see Figure B3). Despite having some different results, depending on the data source (Appendix B), we are using the demand function from PEAG [2008] due to the better performance of the empirical results under these parameters.[12]

In the case of the supply or pumping cost function, the literature shows that there is a large variation in the pumping costs in Spain. Llamas and Garrido [2007] established that the pumping costs in Spain vary between 0.03 and 0.30 €/m^3, with a Spain-wide average of around 0.12 €/m^3. Some studies suggest that in the Guadiana basin in Spain the total average financial costs of groundwater are around 0.10 €/m^3 (Blanco-Gutiérrez et al. [2011]). Garrido et al. [2005] estimated that groundwater irrigated productivity in Spain ranges between 0.20 and 4.00 €/m^3 (depending on the crop type); and the pumping costs represent between 2% and 15% of this productivity. In the case of WLM aquifer, Garrido and Calatrava [2010] estimated that groundwater pumping costs were around 0.08 €/m^3. The electricity costs to pump the water from the WLM aquifer are estimated at 0.0004 €/m^3

TABLE 1. Values of parameters used for the analysis of the Western la Mancha aquifer and the Tablas de Daimiel wetland.

Parameters	Description	Units	Value
K	Water demand slope	€/Mm^3	0.7272
G	Water demand intercept	€/Mm^3	726.71
C_0	Pumping costs intercept	€/Mm^3	319,500
C_1	Pumping costs slope	€/Mm^{3}m	500
α	Return flow coefficient	–	0.2
H_0	Current water table	m	640
R	Natural recharge	Mm^3	360
A	Aquifer area	ha	550,000
S	Storativity coefficient	–	0.023
S_L	Surface elevation	m	665
R	Social discount rate	%	0.02
σ	Ecological function intercept	ha	2085
ρ	Ecosystem function slope	ha/m	75.82
ξ	Economic value of ecosystems	€/ha	720

(Esteban and Albiac [2011]). Some authors have estimated that fix pumping costs (such as extraction technology amortization) are higher than 0.08 €/m^3 (MIMAM [2003], PEAG [2005]). Using the previous information and following equations (5) and (6), we have approximated the pumping cost function coefficients as $C_0 = 319,000$ €/Mm^3 and $C_1 = -500$ €/Mm^3.[13]

The total value of the wetlands is difficult to estimate due to the large number of ecosystem services that it provides, and also because most services do not have a proper market price. A growing body of literature calculates the value of different wetlands; one of the main characteristic of this literature is the extreme differences in the values estimated due to wetland types, good or service analyzed, and valuation method used (Brander et al. [2006]). Groot et al. [2006] summarize several of the ecosystem services' values provided by wetlands. The literature provides values ranging typically between 1000 and 10,000 €/ha (Costanza et al. [1997]). However, there are some examples in the literature with higher values, up to 60,000 €/ha (Hanemann et al. [1990], Kosz [1996], Ledoux [2003], or Moeltner and Woodward [2009]).

We have estimated the parameters of the ecosystem function as a measure of the Tablas de Daimiel flood area reduction due to aquifer depletion (see Appendix C). The economic value of the ecosystem is taken from Judez et al. [2000]. These authors estimated the total recreational value of Tablas de Daimiel at 538,510 €.[14] This value has been adjusted to the 1996 flood area, of around 750 ha. Therefore, the total recreational value of Tablas de Daimiel is set at 720 €/ha. Since the economic value of Tablas de Daimiel does not include all possible benefit values that presumably represent additional services observed in the literature (Groot et al. [2006]), we use the 720 €/ha as a lower bound. Then we apply a sensitivity analysis with higher values of wetland services as is reported below.

The estimation of the ecosystem function is presented in Appendix C. The slope of the wetland health function varies, depending on the specific population and the type of wetland. An additional sensitivity analysis with different values of the ecosystem's function slope is also performed since the value of the slope is subject to high uncertainties. In the sensitivity analysis we use lower and higher slope values than the one estimated.

4.3. Empirical results. The numerical results are summarized in Table 2. The optimal values of the water table and the extractions are reported both under the social planner regime (all externalities are accounted for, Section 3.2) and also under the private optimal control regime (only nonenvironmental externalities are internalized, Section 3.2). Table 2 also presents the results under the sensitivity analysis scenarios: sensitivity analysis 1, which refers to an economic value of the ecosystems higher than the base (3280 €/ha); and sensitivity analyses 2 and 3 that refer to ecosystems' response function slopes that vary between 37.91 ha/m and 151.64 ha/m, respectively.[15]

The empirical results illustrate how the inclusion of environmental externalities in the social planner optimization of groundwater management problem leads to lower levels of groundwater extractions, and then, an increase in the water table levels. This effect is larger the higher is the economic value of the ecosystems and the higher is the ecosystems' function slope (see sensitivity analyses 1 and 3).

Following what the theoretical results suggested (equations (27) and (28)) the results in Table 2 show that when GDEs are taken into account, groundwater management is a necessary policy aimed to reduce extractions and achieve higher groundwater table levels. However, this is not always true (see, for example, sensitivity analysis 2). If ecosystems present large resilience to habitat changes (represented by a lower ecosystem health function slope) the results under the social planner and the private problem are not as different. This can even not justify regulation due to the intervention transaction cost.

Figures 4 and 5 illustrate the behavior of the water extractions and the water table level over a period of 450 years and under all scenarios. It is seen in these figures that both the extraction and the water table levels are sensitive to the slope of the ecosystem response function and to the value of the ecosystem. When the value of the ecosystem is large (3280 €/ha) or when the ecosystems' function slope

TABLE 2. Summary of the optimal levels of groundwater extractions and water table level in WLM.

Private optimal control regime (Nonenvironmental externalities accounted: Gisser and Sánchez model)
$W(t) = 450 + \{122.98\} \cdot e^{-0.01366 \cdot t}$ $H(t) = 639.66 + \{0.34\} \cdot e^{-0.01366 \cdot t}$
Social planner regime[a] (Environmental externalities accounted)
$W(t) = 450 + \{-2.52\} \cdot e^{-0.01366 \cdot t}$ $H(t) = 640.006 + \{-0.006\} \cdot e^{-0.01366 \cdot t}$
Social planner regime: Sensitivity analysis 1 (Social planner regime: 3280 €/ha)
$W(t) = 450 + \{-330.19\} \cdot e^{-0.01366 \cdot t}$ $H(t) = 640.91 + \{-0.91\} \cdot e^{-0.01366 \cdot t}$
Social planner regime: Sensitivity analysis 2 (Social planner regime: 37.91 ha/m)
$W(t) = 450 + \{-110.43\} \cdot e^{-0.01366 \cdot t}$ $H(t) = 639.70 + \{0.30\} \cdot e^{-0.01366 \cdot t}$
Social planner regime: Sensitivity analysis 3 (Social planner regime: 151.64 ha/m)
$W(t) = 450 + \{-125.29\} \cdot e^{-0.01366 \cdot t}$ $H(t) = 640.34 + \{-0.34\} \cdot e^{-0.01366 \cdot t}$

[a] Baseline social planner regime with reference values: $\xi = 720$ €/ha; $\rho = 75.82$ ha/m.

is high (151.64 ha/m) the results show that farmers should pump much smaller groundwater quantities (or even stop pumping).

The sensitivity analysis corroborates that when a higher economic value of the ecosystem is used (sensitivity analysis 1) the differences between the optimal solutions under the social planner regime and under the private optimal control regime are significant. Similar results are observed in the case of the ecosystem health function slope (sensitivity analysis 3). The lower the ecosystem health function slope, the lower the impact on the optimal solutions (sensitivity analysis 2). And the higher the ecosystem health-function slope, the higher the impact on the optimal solutions (sensitivity analysis 3). As suggested by the theoretical model (Propositions 1 and 2), both the ecosystem economic value and the ecosystem health function slope affect the optimal groundwater management.

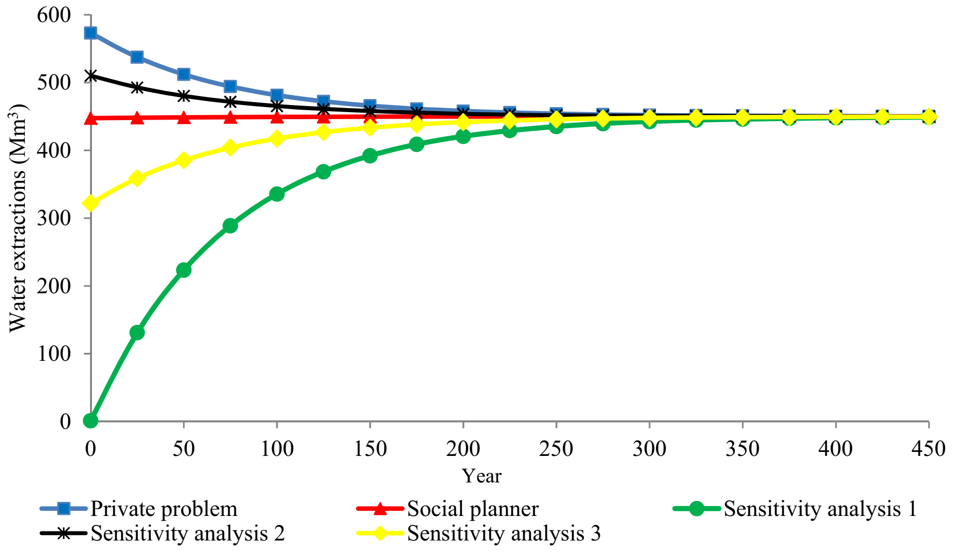

FIGURE 4. Optimal extraction levels under the different scenarios (private and social planner solutions and sensitivity analyses).

Note: Sensitivity analysis 1 (value of ecosystem 3280 €/ha); sensitivity analysis 2 (low value of damage function slope); sensitivity analysis 3 (high value of damage function slope).

The model results suggest that the aquifer reaches a steady state after 94 and approximately 300 years[16] for the social planner and the private profit solutions (Table 3, bottom line). Our simulations suggest that the system stabilizes earlier depending on the simulation. This indicates that the internalization of the GDEs into the groundwater management affects the timing of reaching the steady state. The inclusion of GDEs (social planner regime) causes the steady state to be reached significantly earlier compared to the private problem regime; nevertheless, in the simulations where ecosystems have a higher slope and/or higher economic value (sensitivity analyses 1 and 3) the differences are not significant. This suggests that the results are sensitive to the parameters used. However, the differences in the equilibrium timing involve some policy implications not only because the results for the water table and the extractions are different depending on the internalization of GDEs but also because there are differences in the timing to stabilize the system. While the average steady state level in our model is reached after 300 years, which may sound quite long, we find that in many other cases reported in the literature reaching a steady state after 250–500 years (and more) is not unheard of (e.g., Brill and Burness [1994], Knapp and Olson [1995], Burness and Brill [2001], Knapp and Baerenklau [2006] and Gisser and Sánchez [1980]).

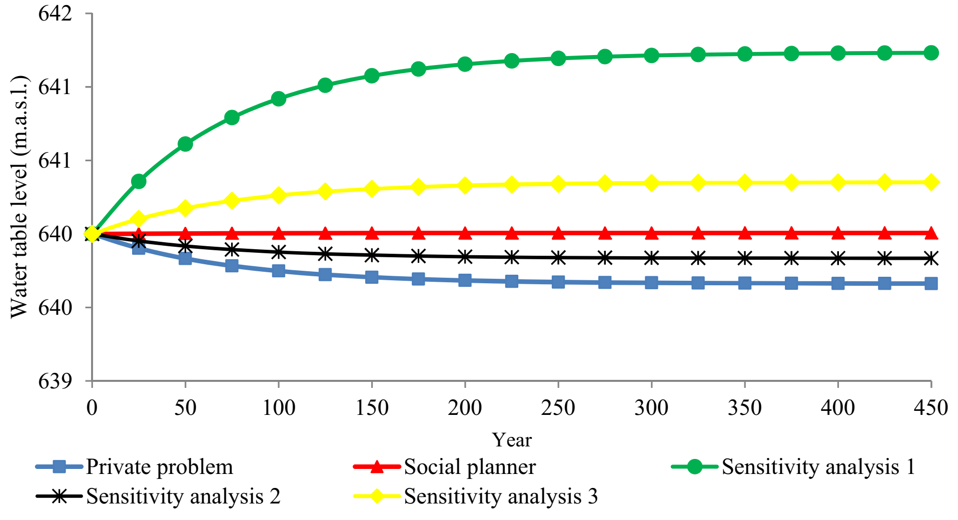

FIGURE 5. Optimal water table levels under the different scenarios (private and social planner solutions and sensitivity analyses)

Note: Sensitivity analysis 1 (value of ecosystem 3280 €/ha); sensitivity analysis 2 (low value of damage function slope); sensitivity analysis 3 (high value of damage function slope).

TABLE 3. Net present value of social welfare, private profits, and ecosystems.

	Private problem	Social planner	Sensitivity analysis 1	Sensitivity analysis 2	Sensitivity analysis 3
Total social welfare (million €)	52.34	56.58	102.46	56.74	56.87
Farmers' private profits (million €)	45.83	43.51	28.68	44.77	40.38
Ecosystem economic value (million €)	6.51	13.07	73.78	11.97	16.49
Ecosystems' flood area (ha)[a]	164.23	190.01	281.68	177.13	215.80
Time to reach steady state	339	94	357	401	286

Note: The results are the aggregate value for the 300 years (approximate year when the steady state of the aquifer system is reached for all five scenarios).
[a] Area reached in year 300.

To better support the empirical results we have included Table 3 that contains a summary of the aggregate values (by year 300 which is the year by which a steady state of the optimal solution is reached) of the net present value of social welfare, farmers' private profits, and ecosystem economic value for all scenarios. The results show how the highest values of social welfare are reached under the scenarios with the highest economic value of the ecosystems and the highest ecosystem health function slope (sensitivity analyses 1 and 3). However, these scenarios are also associated with the lowest private profits. The results clearly show how in the scenarios where GDEs are not taken into account, or are less important (sensitivity analysis 2), both the economic value of the ecosystems and the ecosystems flood area (status) are the lowest.

Two additional clarifications are necessary. The empirical results of this model unquestionably replicate the theoretical solutions. However, it is important to mention that the movement of the water table toward a steady state is quite slow and not exactly corresponding to reality. When large reductions in groundwater extractions (sensitivity analyses 1 and 3) occur, a further increase in the water table level should be expected and also higher values of social welfare and ecosystems could be expected. Possible explanations for these effects are the fact that some of the parameters are approximations and we face a loss of accuracy in the model. On the other hand, neither the demand functions, the pumping costs, nor the ecosystem function are linear ones. The use of nonlinear approaches, for all the functions, included the ecosystems health function, could yield more realistic results.

5. Conclusions. GDEs are ecological systems connected to aquifers that serve as their main water source. These ecosystems provide an important stream of goods and services to society, most of which do not have a proper market price. Traditionally, such ecosystems are not taken into account in groundwater management plans and thus are subject to stress and damage, which then inflicts back on society.

There is mounting evidence that significant depletion and deterioration of groundwater bodies around the world threatens the survival of many of these ecosystems. Such threats are exacerbated by likely impacts of climate change on the aquifers' natural recharge.

This paper proposes a groundwater management model in which a social planner internalizes the social benefits of the GDEs. The social planner maximizes the sum of private benefits of groundwater users and the social benefits provided by GDEs. The paper highlights that an efficient groundwater management policy requires a good knowledge of the GDEs interaction processes with the aquifer water.

Ecosystems behave in a different manner depending on their intrinsic characteristics and this fact affects the optimal regulation of the aquifer. The theoretical and empirical results suggest that both the economic value of ecosystems and the ecosystem health function slope affect the optimal extraction rates and the optimal water table level. When ecosystems produce goods and/or services with a large monetary value or when the absolute value of the slope of the ecosystem health function is large, then groundwater management is essential for protecting them and their services to society.

The results demonstrate both theoretically and empirically the importance of incorporating the environmental externality in social analyses, but also the importance of taking into account the specific ecosystem functions in such analysis. Optimal results can largely change depending on the type of ecosystem behavior. This result justifies investment in acquiring better knowledge of ecosystem processes and their relationship with the rest of the environment.

Several caveats apply to our results. First, we recognize the fact that our model parameters are subject to great uncertainty, due to scientific deficiency regarding the ecosystem sensitivity to water stress. While we used a range of values based on the literature available to us, we still believe that additional studies are necessary. Second, we are aware of the simplification we introduced to the model by linearizing all the functions used. Additionally, the derived demand for water in the WLM region, and the relationship between water table depletion and the area of the Tablas de Daimiel are estimations and approximations using available data. Our future research would also focus on the impacts of noncontinuous linear specifications and nonlinearity forms of the ecosystem health function, and also on their empirical estimation in the numerical model.

The fact that the model reaches steady state at such late period (average of 300 years) could introduce a political issue that may reduce the acceptability of the various intervention policies. Therefore, the difference in the timing of reaching the steady state is also an important issue to take into account in policy interventions. While we see this result to be an empirical issue, we still think that it raises an interesting policy question that reminds the debate on climate change mitigation efforts that will show impact in the distant future, with all implications to acceptability of such mitigation policies.

Acknowledgments. This paper was initiated while the first author was a postdoctoral researcher at the Water Science and Policy Center and the Department of Environmental Sciences, University of California, Riverside. The authors want to thank Darrel Jenerette for his valuable help in the ecosystem behavior function description. We are grateful to two anonymous referees and the editor of the journal for constructive comments.

Appendix A

Proof of propositions.

Proof of Proposition 1. The derivative of the water table level and the level of extractions with respect to the economic value of the ecosystem show the impacts of the value of the ecosystems on the optimal model solutions

$$\frac{\partial W(t)}{\partial \xi} = -\frac{(\alpha-1)\cdot k\cdot \rho}{r\cdot AS}\cdot e^{tx_2}, \tag{A1}$$

$$\frac{\partial H(t)}{\partial \xi} = \frac{(\alpha-1)\cdot \rho}{r\cdot AS\cdot C_1} - \frac{(\alpha-1)\cdot \rho}{r\cdot AS\cdot C_1}\cdot e^{tx_2}. \tag{A2}$$

In the case of the level of extractions (equation A1), and knowing that $(\alpha-1)<0$, $k<0, r>0$, and $\rho>0$; we can conclude that the sign of the derivative is negative. This means that when the value of the ecosystem increases, the level of the optimal extractions decrease.

In the case of the water table level (equation A2), and knowing that $C_1<0$; the sign of the derivate is positive.[17] This means that as long as the economic value of the ecosystem increases, the optimal water table also increases. ■

Proof of Proposition 2. The derivative of the water table level and the level of extractions with respect to the ecosystems' function slope show the impacts that the ecology of the ecosystems has on the optimal solutions. ■

$$\frac{\partial W(t)}{\partial \rho_i} = -\frac{(\alpha-1)\cdot k\cdot \xi}{r\cdot AS}, \tag{A3}$$

$$\frac{\partial H(t)}{\partial \rho_i} = \frac{(\alpha-1)\cdot \xi}{r\cdot AS\cdot C_1} - \frac{(\alpha-1)\cdot \xi}{r\cdot AS\cdot C_1}\cdot e^{tx_2}. \tag{A4}$$

For similar reasons as in the case of the economic value of the ecosystem, the derivative with respect to the ecosystem's function slope shows that as the absolute value of the slope increases, indicating more sensitive ecosystem, larger damages to the ecosystem results from a given deterioration of the external conditions. In this case, the water table level (equation A4) increases, and the water extractions (equation A3) decrease in the optimal solution.

Appendix B

Estimation of the water demand function. We estimate irrigation demand function from various relevant sources as described below.

Estimation of the derived water demand using data in Esteban and Albiac [2011]: A dynamic groundwater optimal control model was applied to the condition in the WLM aquifer, using the software GAMS. We applied the procedure suggested in Tsur et al. [2004, chapter 4] to estimate the derived demand for irrigation water. The procedure is applied for different levels of water constraints faced by the irrigators. The constraints impose maximum levels of water extractions, which are increasing by constant increments, starting with 250–750 Mm^3 in the WLM. For each run the shadow price in the equilibrium is recorded. Figure B1 presents the relationship between the shadow prices and the water constraints. A linear version of the derived demand equation for irrigation water (equation B1) was estimated to be

$$W = 4400.73 - 0.097 \cdot P. \tag{B1}$$

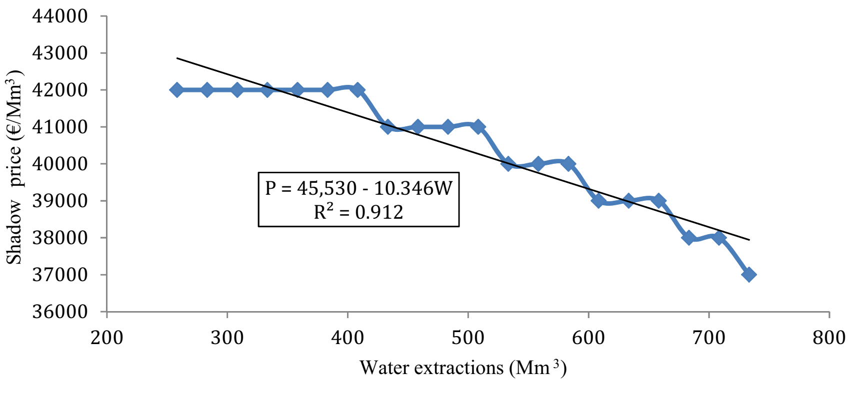

FIGURE B1. Derived demand for irrigation water in WLM.

Source: Based on data in Esteban and Albiac [2011].

Estimation of the water demand from Blanco-Gutiérrez et al. [2011]: Using the data provided by these authors we estimated a groundwater inverse demand function. Figure B2 shows the linear relationship between water extractions (Mm^3) and groundwater shadow price (€/Mm^3).[18]

A linear version of the derived demand equation for irrigation water (equation B2) was estimated to be

$$W = 472.61 - 0.0011 \cdot P. \tag{B2}$$

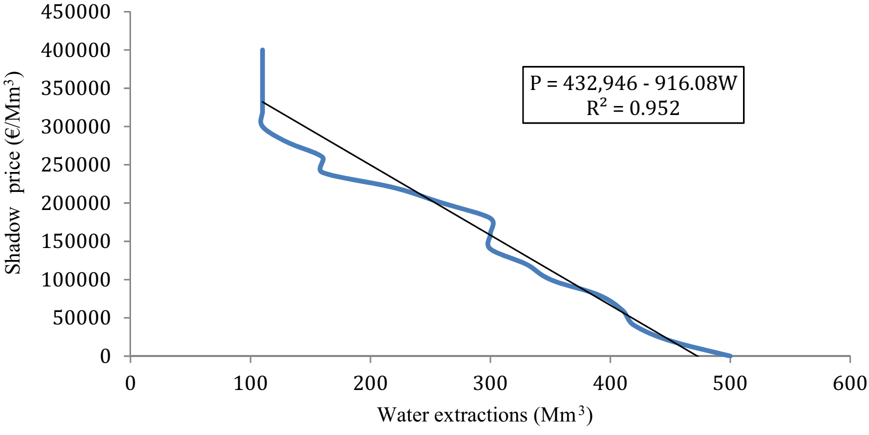

FIGURE B2. Derived demand for irrigation water in WLM.

Source: Based on data in Blanco-Gutiérrez et al. [2011].

Estimation of the water demand from PEAG [2008]: We estimated a groundwater inverse demand function. Figure B3 shows the linear relationship between water extractions (Mm^3) and groundwater shadow price (€/Mm^3). The groundwater demand equation is

$$W = 726.71 - 0.7272 \cdot P. \tag{B3}$$

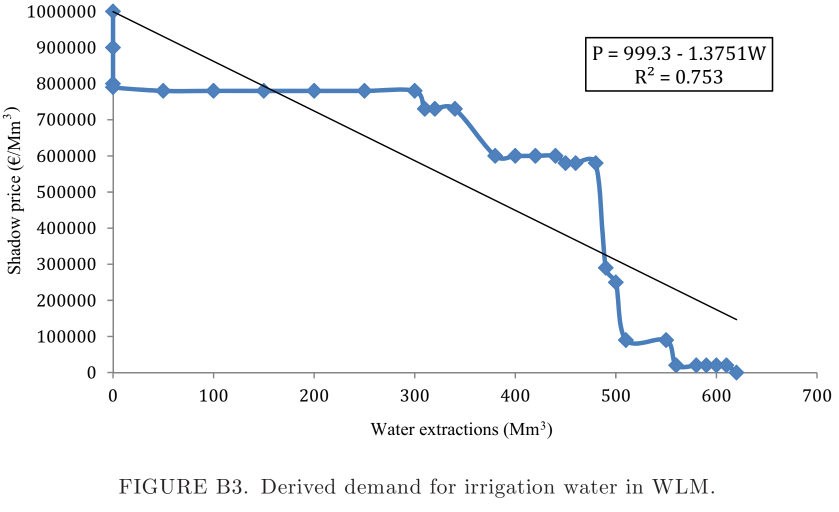

FIGURE B3. Derived demand for irrigation water in WLM.

Source: Based on data in PEAG [2008].

Estimation of the water demand from Rubio and Castro [1996]: The authors estimated the groundwater demand function in WLM with the resulting parameters of $k = -0.001$ and $g = 244$. The linear water demand proposed by Rubio and Castro considers the existence of additive uncertainty

$$W = 244 - 0.001 \cdot P^{\eta} + u^{\gamma}, \tag{B4}$$

where $\eta, \gamma > 0$.

Appendix C

Estimation of the relationship between water table depletion and area of the Tablas de Daimiel. The ecosystem health function is calculated with data from a time series of the WLM aquifer depletion (Martínez-Santos et al. [2008]) and from the evolution of Tablas de Daimiel flood area (Ruíz de la Hermosa [2011]). By regressing the hectares of Tablas de Daimiel against the water table depletion, the ecosystem's health function is obtained in terms of size of the wetland (Figure C1).

The estimated linear regression corresponds to equation (8). The estimated ecosystem's status function is

$$D\left(H\right) = 2085 - 75.82 \cdot \left(S_L - H\right). \tag{C1}$$

Wetlands ecosystems are mainly affected by rainfall levels (Figure C2). In wet years the wetland flood area is large; and during dry years the flood area of wetlands greatly decreases. However, the groundwater level also affects the flood area of the wetland system. We can observe (Figure C2) how the wetland flood area in the case of Tablas de Daimiel is being significantly affected by the depletion of the aquifer due to large extractions for irrigation. The decrease in the water table in the past 20 dry years led to shrinking of the wetland, which in turn created significant and even irreversible damages to several of its subecosystems.

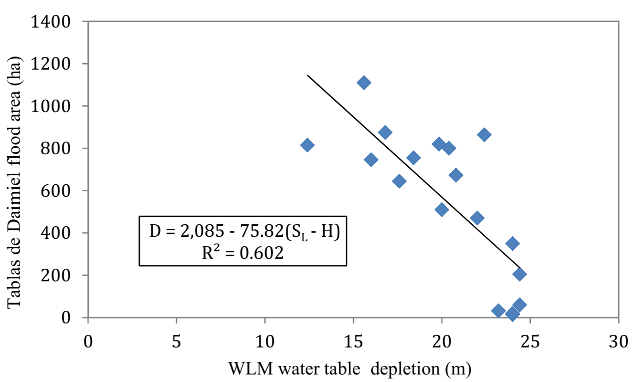

FIGURE C1. Function of the flood acreage in Tablas de Daimiel with respect to the WLM aquifer depletion.

Note: Y axis is Tablas de Daimiel flood area. X axis is WLM water table depletion.

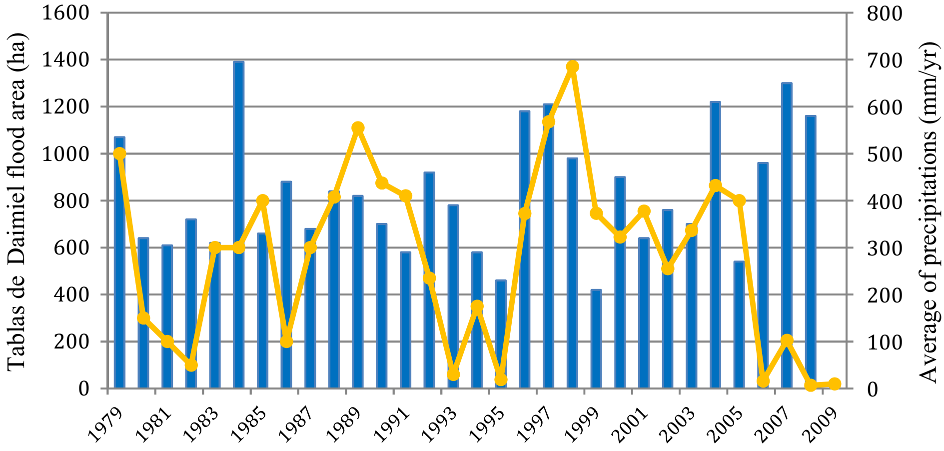

FIGURE C2. Representation of the Tablas de Daimiel evolution and the average rainfall in the area.

Note: The flood area of Tablas de Daimiel is represented by the line, and annual precipitation is represented by the bars.

ENDNOTES

1. We use the terms "health" and "status" interchangeably. Both terms are related to the conditions of the ecosystems and can be measured under diverse parameters as the number of species, population size, productivity level, etc.

2. These authors have reviewed several papers reporting nonlinear changes in ecosystems.

3. For further details about shifts in ecosystems' behavior due to change in their habitat see Scheffer et al. [2001]. In this paper, we use the terminology suggested by these authors.

4. In this paper, the slope and the intercept of the ecosystem health function show how gradual or abrupt is the change in the GDE status due to modifications in their natural habitat (e.g., the decrease in the aquifer storage or the deterioration in the quality of the groundwater).

5. The storativity coefficient depends on the physical properties of the aquifer and indicates the capacity of an aquifer to release the groundwater stored in it.

6. If there is no depletion in the aquifer and $S_L = H$, then the status or ecosystem health is at maximum.

7. Gisser and Sánchez [1980] denote this equilibrium as the optimal control, where regulation exists and extraction externality is internalized. Under the other regime defined by these authors (farmers' private problem) farmers do not even internalize the impact that their groundwater extractions cause to other farmers, ignoring the extraction externality.

8. For simplicity, the notation in our model follows the one proposed by Gisser and Sánchez [1980: 640].

9. For further information about the simplification and final solution of the model see Gisser and Sánchez [1980].

10. In their model, Gisser and Sánchez assumed that, due to the large size of the aquifer, the expression in equation (29) could be simplified to: $x_2 \cong \frac{k \cdot C_1 \cdot (\alpha - 1)}{AS} < 0$.

11. See Esteban and Albiac [2011].

12. We have estimated the model under all sources of the demand parameters. The outcomes corroborate the theoretical results of this paper; however, the movement of the water table and/or extraction rates when using the demand parameters from all but PEAG [2008] was not coherent with what could be expected in reality. We do not incorporate these results due to space issues; however, we will provide these results upon request.

13. We have increased the electricity costs of water pumping from 0.0004 to 0.0005 €/m^3 due to the tariff increases in 2010. The fixed pumping costs (C_0) in this model are approximated to obtain reasonable results of farmers' profits in the study area.

14. Judez et al. [2000] estimated the recreational value of Tablas de Daimiel in 1996, using contingent valuation and travel cost methods.

15. In sensitivity analysis 1 we use an ecosystem value that decreases groundwater extractions to almost zero. In the case of sensitivity analyses 2 and 3 we use a value of the slope that is half and double the estimated value, respectively.

16. This is an average of the year of reaching the steady state across all simulations. Furthermore, even realizing that the steady state is reached in different years, after year 300 the differences in the results, for both the water table and the extractions, between this year and the equilibrium are negligible in all simulations (e.g., 1 mm of difference in the water table).

17. Realize that $\frac{(\alpha-1)\cdot\rho}{r\cdot AS\cdot C_1} > \frac{(\alpha-1)\cdot\rho}{r\cdot AS\cdot C_1}\cdot e^{tx_2}$ because x_2 is always <0 (see equation (29)).

18. Blanco-Gutiérrez et al. [2011] relate the quantity of water pumped in Mm^3 with the water price in €/m^3. In order to maintain a coherence in the paper's units we have converted the water prices to €/Mm^3.

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