Agricultural pollution control under Spanish and European environmental policies

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[1] Nonpoint pollution from agriculture is an important environmental policy issue in Spain and the European Union. Agricultural pollution in Spain is being addressed by the National Irrigation Plan and by the European Water Framework Directive. This article contributes to the ongoing policy decision process by analyzing nonpoint pollution control and presenting results on the efficiency of abatement measures. Results question the reliance of the Water Framework Directive on water pricing as a pollution instrument for reaching good status for all waters because higher water prices close to full recovery cost advocated by the directive appear to be inefficient as an emission control instrument. Another important result is that abatement measures based on input taxes and standards on nitrogen appear to be more suitable than the National Irrigation Plan subsidies designed to promote irrigation investments. The results also contribute with further evidence to the discussion on the appropriate instrument base for pollution control, proving that nonpoint pollution control instruments cannot be assessed accurately without a correct understanding of the key underlying biophysical processes. Nonpoint pollution is characterized by nonlinearities, dynamics, and spatial dependency, and neglect of the dynamic aspects may lead to serious consequences for the design of measures. Finally, a quantitative assessment has been performed to explore discriminating measures based on crop pollution potential on vulnerable soils. No significant welfare gains are found from discriminating control, although results are contingent upon the level of damage, and discrimination could be justified in areas with valuable ecosystems and severe pollution damages. INDEX TERMS: 1803 Hydrology: Anthropogenic effects; 1842 Hydrology: Irrigation; 1871 Hydrology: Surface water quality; 6304 Policy Sciences: Benefit-cost analysis; KEYWORDS: nonpoint pollution, Water Framework Directive, National Irrigation Plan, soil nitrogen dynamics, efficiency of abatement measures


1. Introduction

[2] Water resources nonpoint pollution from agriculture is becoming an important environmental policy issue in Spain, a southern European country with a large irrigated agricultural sector, which covers 3.5 million ha and consumes 24,100 hm³ of total 30,400 water demand. The policy issue has been addressed by both, domestic and European Union legislation. While the Spanish government has enacted the National Hydrological Plan and the National Irrigation Plan, the European Union (EU) has enacted the Water Framework Directive.

[3] The National Hydrological Plan involves significant investments aimed at increasing the water supply for irrigation, urban and industrial users. The key project of the Hydrological Plan is the Ebro interbasin transfer, from northeastern to southeastern Spain, designed to solve the severe degradation of water resources in the receiving basins, although this supply increasing approach has been criticized by some experts. Economic arguments against the transfer are given by Albiac et al. [2003], Howitt [2003], and Hanemann [2003], and environmental arguments dealing with the degradation of the Ebro Delta and the fluvial and marine ecosystems are given by Ibañez and Prat [2003] and Prat and Ibañez [2003]. The National Irrigation Plan subsidizes the modernization of the largely outdated irrigation facilities, in order to save water resources and improve the competitiveness of agricultural production. Up to the 2008 horizon, planned investments are 19 billion euro under the National Hydrological Plan, and 5 billion euro under the National Irrigation Plan which covers 1.1 million ha (1 euro = 1.2 US dollars).

[4] The European Water Framework Directive protects all continental, coastal and subsurface waters, and is aimed at improving water quality and ecosystems conditions, promoting the sustainable use of water, and reducing emissions and discharges to water media. The directive is aimed at
securing a good quality water supply while reducing water pollution, and is based on the principles of river basin management and public participation. Water pricing should approximate full recovery costs to increase water use efficiency, and costs should include extraction, distribution and treatment costs, environmental costs, and resource value costs. The directive introduces a combination of emission limits and water quality standards, with deadlines to achieve good status for all waters.

[5] These domestic and European policy initiatives will have a strong impact on water nonpoint pollution in Spain. The National Hydrological Plan consist mainly in investments to expand water availability, and may have negative effects on nonpoint agricultural pollution because of the expansion of irrigation in Mediterranean coastal agriculture. In this region, fruits and vegetables are cultivated under intensive production technologies, which include a substantial greenhouse acreage. The National Irrigation Plan, on the other hand, is destined to modernize irrigation technologies in the more traditional irrigation areas of continental Spain, although the empirical evidence is ambiguous on the effects on water extraction savings and nonpoint pollution abatement. These effects depend basically on implementing a tradeoff between public subsidies and reductions in water concessions, an aspect which has not been considered in the plan [Sumpsi et al., 1998; Uku, 2003].

[6] The European Water Framework Directive also has a great potential to abate nonpoint pollution in Spain, and this initiative is supported by the findings of the European Environmental Agency, which point to agricultural nonpoint pollution as the primary cause of water quality deterioration in many European watersheds [European Environment Agency (EEA), 1999]. However, the reliance of the directive on water pricing to curb demand may lead to its failure in southern countries such as Spain, with high irrigation demand and quality problems compounded by water scarcity.

[7] Water pricing will not solve scarcity or improve quality in the more degraded areas, because rising water prices would reduce consumption in large irrigation districts of inland Spain, based on collective systems and low-profit crops, where degradation problems are moderate. However, water demand will not respond to higher prices in Mediterranean areas, reliant on individual extractions from aquifers and high-profit crops, where pressure on water resources is pervasive and degradation is severe [Massarutto, 2003]. Water pricing fails as a workable policy for curbing demand in Mediterranean regions for several reasons. The first is that, after decades of mismanagement, the number of illegal private wells is huge and there is no control over the volume pumped from either legal or illegal wells. A second reason is related to the water price level that is needed to curb demand. Shadow prices of water in areas under greenhouse production can reach three to five euros per cubic meter, against 10–20 cents €/m³ in inland Spain, while current water prices in the irrigation zones of the Mediterranean are between 6 and 21 cents/m³ compared to 2–5 cents €/m³ in inland collective irrigation systems [Ministerio de Medio Ambiente, 2000; Albiac et al., 2004]. With urban prices in Spain close to or below one euro per cubic meter, and seawater desalination at around 50 cents €/m³, it would seem unacceptable to set agricultural prices in water scarcity areas above urban and desalination prices. Though a policy designed to control aquifer overdraft would be quite difficult to implement, a water pricing policy that were to drive prices above the three to five euros shadow price per cubic meter for private extractions would be impossible to implement, both because of its technical and administrative unfeasibility and the daunting prospect of social opposition from farmers. These more degraded areas therefore require other directive instruments, such as controlling aquifer overdraft by reducing concessions, and enforcing ambient quality standards and pollution emissions limits.

[8] Nonpoint pollution from agriculture in Spain has increased over recent decades, as a consequence of large-scale investments in production-enhancing technologies that involve the intensive use of machinery, industrial fertilizers and pesticides. These technologies have increased yields but have also resulted in water quality and soil degradation problems. One of the main problems of nonpoint source pollution is originated by nitrate fertilizers, which have negative effects on water quality, damaging ecosystems and posing health risks to humans. Nitrogen fertilizer use per hectare in Spain is close to the EU average (70 kg/ha), and the nitrogen surplus in soils in Spain is estimated at 40 kg/ha, well below the 215 kg/ha surplus in Netherlands or the 100 kg/ha surplus of Belgium and Germany [EEA, 2003]. However, excessive fertilizer application in Spain is a serious problem in some regions with irrigation-intensive agriculture such as the Mediterranean coast, and several inland watersheds in the Castilla-La Mancha region and the Ebro and Guadalquivir basins, with concentrations between 50–100 mg N03/L.

[9] The new Water Framework Directive introduces drastic changes in the control, evaluation and management of water resources, and requires the achievement of good ecological status and good ecological potential of water bodies across the EU by 2015. To achieve this good status, the emphasis of current legislation on controlling point sources must be expanded to cover nonpoint pollution sources from agriculture [EEA, 2003]. There appears to be some lack of consistency between the nonpoint pollution abatement goal of the European Directive, and the goals of the Spanish National Hydrological and National Irrigation Plans. As indicated above, the main objective of the National Hydrological Plan is the expansion of water supply to urban, industrial and agricultural users, and the additional water supply of 600 hm³ to Mediterranean agriculture, will surely increase nonpoint pollution. The effects of the National Irrigation Plan on nonpoint pollution are more ambiguous, as will be explained later in the results section, because investments to modernize irrigation technologies may induce the cultivation of high-profit crops, which are more input intensive and generate more pollution.

[10] The purpose of this study is to contribute to the ongoing policy discussion in the EU, by evaluating control measures for nitrogen pollution that are being considered in the design of policy measures. The analysis focuses on nitrate pollution from irrigation, and the goal is to rank alternative measures for the abatement of nitrogen pollution by their cost efficiency. Several studies have examined the cost efficiency of alternative measures for some crops: corn [Yadav, 1997; Vickner et al., 1998], wheat and barley [Miettinen, 1993; Schmid, 2001], corn, wheat, barley, rice,
sunflower and alfalfa [Martinez, 2002], and cotton [Khanna et al., 2000]. Ranking of measures is contingent not only upon crop type but also upon soils, since soil type may have a significant impact on the cost efficiency of measures. Information on possible impacts is scarce and limited to corn and alfalfa with three soil classes [Thomas and Boisvert, 1994], lettuce with two soils [Helfand and House, 1995], and corn with three soils [Martinez and Albic, 2004].

[11] The present study emphasizes the importance of aspects such as properly understanding the biophysical processes in nonpoint pollution for a correct assessment of the efficiency of instruments, and in particular the inclusion of dynamics in nonpoint pollution. The later issue is discussed in the methodology section, with some examples from the literature on nonpoint pollution that suggest that the neglect of dynamics may lead to the wrong choice of the instrument base.

[12] A correct understanding and representation of the physical processes involved requires knowledge about the main factors driving nitrogen emissions and their inter-relationships. The main factors are input use, crop type, irrigation technology, and soil characteristics and their links through the crop production and pollution functions. Some local studies on the selected irrigation area, the Flumen-Monegros district in the Ebro basin of the Iberian Peninsula, highlight the importance of these factors in evaluating pollution abatement measures [Feijoo et al., 2000; Martinez et al., 2002; Martinez and Albic, 2004; Uku, 2003].

2. Methodology

[13] The economic theory of nonpoint pollution control was initiated by Griffin and Bromley [1982] and Shortle and Dunn [1986], who proposed instruments based on the measurement of inputs or practices, and by Segerson [1988], who proposed instruments based on the measurement of ambient pollution. The economic analysis is based on three functions describing nonpoint pollution, ambient pollution and environmental damage costs. In the nonpoint production function, pollution emissions depend on production and pollution control choices, and farm site characteristics. The ambient pollution function is driven by nonpoint pollution and watershed characteristics. Finally, the damage costs function relates environmental costs to ambient pollution [Horan and Shortle, 2001].

[14] The problem is then formulated as the maximization of the difference between profits derived from polluting activities (crop production), and pollution costs. For the case of a nitrogen-polluting input $n$, $\Pi_k(n)$ defines the farm’s profit function restricted on the nonpoint-polluting input $n$, and $D[a(\Pi_k(n))]$ is the damage cost function from ambient pollution $a$, which is driven by nitrogen leaching $l$ generated by the nitrogen input $n$. The social net benefit of farm activities is then given by $\text{SNB} = \Pi_k(n) - D[a(\Pi_k(n))]$.

[15] The instruments available for nonpoint pollution abatement are taxes and standards applied to polluting inputs or to pollution emissions. Farmers maximize after-tax profits given by $\Pi_k(n) - t \cdot n$ in the case of a tax $t^*$ on the polluting input, and by $\Pi_k(n) - \tau \cdot l(n)$ in the case of a tax $\tau^*$ on pollution emissions. A standard on the polluting input is set at $n^*$ and a standard on pollution leaching is set at $l^* = l(n^*)$, which are the duals to tax incentives.

[16] A further refinement of this framework is to introduce stochastic elements linked to weather and uncertainty elements linked to limited information, in the process explaining generation, transport and fate of pollutants. Weather stochastic variables and random variables capturing limited information are included in both, the nonpoint production function and the ambient pollution function. Expected social net benefits are then given by expected restricted profits less expected damage costs, and the optimal tax rate on the polluting input or on leaching, is the sum of expected marginal damages plus a risk term [Horan and Shortle, 2001].

[17] Summing up, there are several possible strategies open to empirical research. One is to emphasize the correct understanding and representation of physical processes involved in pollution formation (nitrogen leaching), or in pollution transport and fate (ambient nitrogen concentration). This strategy will lessen the importance of the random variables representing lack of knowledge on physical processes. A second strategy is to introduce random variables for imperfect information on physical processes, and weather stochastic variables that affect pollution formation, transport and fate, and then analyze the policy measures within an uncertainty framework. This study has focused on the nitrogen leaching process, and not on nitrogen transport and fate. The reasons are that a correct understanding of the biophysical processes in nitrogen leaching is needed to design policies for nitrogen pollution abatement, and also because technical information on transport and fate is at present very scarce in Spain. No weather stochastic variables are included in the production and pollution functions, because stochastic weather is less important in irrigated agriculture.

[18] A static model for several crops and polluting inputs will maximize quasi-rent less damage costs of nitrogen pollution. Assuming price exogeneity and input separability, which implies specific profit functions for each crop, the unrestricted profit function for the farm is obtained by maximizing quasi-rent:

$$
\Pi(p, r, \lambda_n, \lambda_w) = \sum_{i=1}^{m} \pi_i(p_i, r, \lambda_n, \lambda_w) \\
= \max_{x_n, r_n, w_i} \left\{ \sum_{i=1}^{m} \left[ p_i y_i(x_i, n_i, w_i) - r_i x_i - \lambda_n n_i - \lambda_w w_i \right] \right\}
$$

(1)

where $p_i$ is the vector of prices $p_i$ of each crop $i$ ($i = 1, \ldots, m$), $\lambda_n$ and $\lambda_w$ are the prices of the nitrogen polluting inputs, nitrogen fertilizer ($n_i$) and irrigation water ($w_i$), which are used in producing crop $i$. $r$ is the price vector of inputs $x_i$ used by crop $i$ other than polluting inputs, and $y_j(x_i, n_i, w_i)$ is the production function of each crop $i$, which depends on the vector of inputs $x_i$ and the polluting inputs $n_i$ and $w_i$.

[19] The restricted profit function for the farm is obtained maximizing profits subject to the quantities of the polluting
inputs. If the nitrogen polluting inputs are considered fixed at \( n \) and \( w \), the optimization problem becomes:

\[
\Pi_R(n, w) = \Pi(p, r, n, w) = \max_{n_1, w_1, \ldots, n_m, w_m} \left\{ \sum_{i=1}^{m} \pi_i(p_i, r_i, n_i, w_i) : \sum_{i=1}^{m} n_i = n, \sum_{i=1}^{m} w_i = w \right\}
\] (2)

[20] As stated above, the social net benefit of the farm activities is given by \( SNB = \Pi_R(n, w) - D[n(w), \omega] \), the difference between the farm’s profit function restricted on polluting inputs and the damage cost function from nitrogen leaching. Farmers maximize after-tax profits which are given by \( \Pi_R(n, w) - t_g \cdot n - t_w \cdot w \) in the case of taxes \( t_g \) and \( t_w \) on the polluting inputs nitrogen fertilizer and irrigation water, and by \( \Pi_R(n, w) - \tau \cdot h(n, w) \) in the case of a tax \( \tau \) on pollution leaching. Standards on the polluting inputs are set at \( n^* \) and \( w^* \), and a standard on pollution leaching is set at \( \tau^* = \{\tau(n^*, w^*)\} \).

[21] Dynamic aspects are a key feature in nitrogen biophysical processes, and neglect of dynamics may result not only in an incorrect choice of the tax or standard instrument level, but also in the incorrect choice of the instrument base. The assessment of instrument cost efficiency depends on the accurate understanding and representation of the biophysical processes involved in nonpoint emissions. An example of this could be taken from the work of Horan and Shortle [2001], who discuss the choice of the instrument base to abate nitrate leaching, using the findings by Helfand and House [1995] and Larson et al. [1996]. They indicate that instruments based on irrigation water are more cost-effective than those based on nitrogen use. The reason they give for this is that nitrate leaching seems to be more highly correlated with irrigation water than with nitrogen use, their conclusion being that the correct instrument base is not the nutrient causing pollution but the input more correlated with pollution.

[22] Our empirical results contradict this choice of instrument base, a discrepancy that might be explained by variations in physical, climatic and agricultural characteristics, leading to different choices of instrument base. An alternative explanation, however, might be that nitrogen soil dynamics is a relevant factor requiring consideration. This explanation assumes that when the level of nitrogen input changes, nitrogen content in the soil adjusts over several production periods. The consequence would be that when imposing adjustment over one period, a substantial cutback on fertilizer use is required to achieve a modest reduction in the level of nitrogen leaching. Conversely, continued use of nitrogen fertilizer, combined with a reduction in applied water, will curb nitrogen leaching during the production period, despite the fact that nitrogen is building up in the soil. The issue is further discussed in the results section, where we present our empirical evidence and the empirical results from several studies that support instruments based on nitrogen use.

[23] This study emphasizes the importance of the correct understanding and accurate representation of biophysical processes which drive nonpoint pollution. Nonpoint pollution is characterized by nonlinearities, dynamics and spatial dependency, which have to be incorporated for proper policy instrument design.

3. Dynamic Model

[24] The study analyzes alternative nitrogen pollution control instruments in the Flumen-Monegros irrigation district, located in the Ebro basin of the Iberian Peninsula. A dynamic bioeconomic model is used to rank control instruments by their cost efficiency. The model includes the production and pollution functions of the main crops in the area, and these relationships are estimated using the EPIC crop growth package simulator [Mitchell et al., 1996].

[25] The EPIC package simulates crop growth using local conditions on soils, climate, irrigation water, tillage and other operations. The package is of great use in the study area, because it includes the major local crops and it is designed for environmental analysis. Production and pollution functions for each crop have been estimated, by using the package to generate observations on yields, percolation, runoff, and nitrogen leaching for a range of water and nitrogen input levels. These functions are key components of the bioeconomic model.

[26] Local and technical information has been taken from a large number of primary and secondary sources. Local information includes soil map and crop map cartography [Nogués et al., 1999; Nogués, 2002; Casterad and Herrero, 1998], weather information and data on crop acreage and yields from central and state administrations, tillage operations and input utilization from field surveys, and crop production costs from the Ministerio de Agricultura, Pesca y Alimentación (MAPA) [2000] database. Technical information from experts has been used to run and calibrate EPIC, and the results have been validated with experimental information from field trials, covering both yields and leaching levels.

[27] The nonpoint pollution abatement problem is stated as the maximization of welfare from crop production, and welfare is the sum of quasi-rent from crops and damage costs from nitrogen leaching. When damage costs are internalized, farmers maximize private quasi-rent minus damage costs by selecting the amount of nitrogen fertilizer and irrigation for each crop throughout the time planning horizon \( t = 1, \ldots, 20 \), subject to the dynamics of nitrogen content in the soil, and other constraints representing agronomic, set-aside, and input availability restrictions. In discrete time, the model is as follows:

\[
\max_{x_{i,t}, z_{i,t}} \sum_{i=1}^{6} \sum_{t=1}^{20} \frac{1}{1+r} \left[ \left( p_{i,1} \cdot y_{i,1} - p_{i,1} x_{i,1} - p_{i,0} n_{i,1} - k_{i} + s_{i} \right) - \left( \mu \cdot l_{i,1} \right) \right] \cdot z_{i,1} \]

subject to

\[
g_{i,t+1} = g_{i,1} + n_{i,1} - l_{i,1} - v_{i,1} - u_{i,1} \]

\[
z_{i,t} \leq \sum_{j=1}^{6} z_{j,1-t} \forall \ i, j = 1, \ldots, 6; i \neq j
\]
are yi leaching.

nitrogen damage is defined as a linear function of nitrogen crop revenue minus variable and fixed cost per hectare, and damage costs from nitrogen leaching. Quasi-rent is equal to water applied (m$^3$/ha), case, the magnitude of equal to the cost of removing nitrogen from water. In this process leading to nitrogen pollution.

In the absence of these intertemporal relationships, nitrogen stock in the soil, yields and nitrogen leaching would be representing the dynamic relationships across time periods. In the absence of these intertemporal relationships, nitrogen stock in the soil, yields and nitrogen leaching would be static and constant for all periods. This dynamic aspect is crucial to an accurate representation of the biophysical processes leading to nitrogen pollution.

Crop production and pollution functions are assumed to be quadratic functions of water applied, nitrogen fertilization, and the stock of nitrogen in the soil. The production functions are defined as $y_{i,t} = a_0 + a_1x_{i,t} + a_2x_{i,t}^2 + a_3n_{i,t} + a_4g_{i,t} + a_5g_{i,t}^2$, and the pollution functions are defined as $le_{i,t} = b_{0,t} + b_{1,t}x_{i,t} + b_{2,t}x_{i,t}^2 + b_{3,t}n_{i,t} + b_{4,t}g_{i,t} + b_{5,t}g_{i,t}^2$. The variables corresponding to each crop $i$ are $y_{i,t}$ yield in metric tons per hectare (t/ha), $x_{i,t}$ irrigation water applied (m$^3$/ha), $n_{i,t}$ nitrogen fertilization (kg/ha), $g_{i,t}$ nitrogen stock in the soil (kg/ha), and $le_{i,t}$ nitrogen leaching (kg/ha). Tables 1 and 2 show the estimated production and pollution functions by crop from observations generated with the EPIC package. Other variables in

$$z_{i,t} = \frac{1}{d_i} \sum_{i=1} z_{i,t}, \forall i$$

(6)

$$z_{pact,t} = \sum z_{i,t} \cdot I$$

(7)

$$\sum_{i=1}^{n} a_i \cdot z_{i,t} = M$$

(8)

$$q_{t+1} = \frac{1}{d_t} \sum_{i=1}^{n} g_{i,t+1} = \frac{1}{d_t} \sum_{i=1}^{n} z_{i,t+1}$$

(9)

and then $g_{i,t+1}$ is set equal to $q_{t+1}$.

The objective equation (3) is social welfare through the planning horizon from crop production activities. The objective function has two components, the first term in parenthesis is quasi-rent and the second term $\mu \cdot le_{i,t}$ is damage costs from nitrogen leaching. Quasi-rent is equal to crop revenue minus variable and fixed cost per hectare, and nitrogen damage is defined as a linear function of nitrogen leaching.

Nitrogen damage costs represent nonmarket environmental damages from nitrogen pollution. In the international literature, there is little available information regarding either nonpollution environmental damage costs or the adequate specification of damage cost functions. Since there are no valuation studies for nitrogen pollution damages in Spain, damage costs are approximated by a linear function of nitrogen leaching, where the unit emission cost $\mu$ is set equal to the cost of removing nitrogen from water. In this case, the magnitude of $\mu$ is related to the type of technology applied to remove nitrogen.

The first constraint (4) captures the dynamics of soil nitrogen content, and is defined as the balance of nitrogen entering and leaving the soil, where soil nitrogen in period $t+1$ is equal to nitrogen in period $t$ plus fertilization minus leaching, volatilization and nitrogen uptake by crops. The inclusion of the nitrogen stock in the soil is a key feature for representing the dynamic relationships across time periods. In the absence of these intertemporal relationships, nitrogen stock in the soil, yields and nitrogen leaching would be static and constant for all periods. This dynamic aspect is crucial to an accurate representation of the biophysical processes leading to nitrogen pollution.

Crop production and pollution functions are assumed to be quadratic functions of water applied, nitrogen fertilization, and the stock of nitrogen in the soil. The production functions are defined as $y_{i,t} = a_0 + a_1x_{i,t} + a_2x_{i,t}^2 + a_3n_{i,t} + a_4g_{i,t} + a_5g_{i,t}^2$, and the pollution functions are defined as $le_{i,t} = b_{0,t} + b_{1,t}x_{i,t} + b_{2,t}x_{i,t}^2 + b_{3,t}n_{i,t} + b_{4,t}g_{i,t} + b_{5,t}g_{i,t}^2$. The variables corresponding to each crop $i$ are $y_{i,t}$ yield in metric tons per hectare (t/ha), $x_{i,t}$ irrigation water applied (m$^3$/ha), $n_{i,t}$ nitrogen fertilization (kg/ha), $g_{i,t}$ nitrogen stock in the soil (kg/ha), and $le_{i,t}$ nitrogen leaching (kg/ha). Tables 1 and 2 show the estimated production and pollution functions by crop from observations generated with the EPIC package. Other variables in

**Table 1. Estimated Production Functions for Each Crop**

<table>
<thead>
<tr>
<th>Variable</th>
<th>Com</th>
<th>Barley</th>
<th>Wheat</th>
<th>Sunflower</th>
<th>Alfalfa</th>
<th>Rice</th>
</tr>
</thead>
<tbody>
<tr>
<td>Intercept</td>
<td>-2.77 (12.31)</td>
<td>-0.37 (-1.22)</td>
<td>-0.37 (-1.22)</td>
<td>-1.92 (-1.27)</td>
<td>-0.36 (-1.23)</td>
<td>-0.85</td>
</tr>
<tr>
<td>Water (X)</td>
<td>0.349 x 10^{-2} (5.93)</td>
<td>0.69 x 10^{-2} (6.38)</td>
<td>0.428 x 10^{-2} (5.93)</td>
<td>0.30 x 10^{-2} (6.38)</td>
<td>0.28 x 10^{-2} (6.38)</td>
<td>0.85</td>
</tr>
<tr>
<td>Nitrogen Reserve (G)</td>
<td>0.543 x 10^{-4} (0.44)</td>
<td>0.44 x 10^{-4} (0.44)</td>
<td>0.14 x 10^{-4} (0.44)</td>
<td>0.14 x 10^{-4} (0.44)</td>
<td>0.14 x 10^{-4} (0.44)</td>
<td>0.85</td>
</tr>
<tr>
<td>Adjusted R$^2$</td>
<td>0.83</td>
<td>0.83</td>
<td>0.83</td>
<td>0.83</td>
<td>0.83</td>
<td>0.83</td>
</tr>
<tr>
<td>n</td>
<td>552</td>
<td>352</td>
<td>352</td>
<td>352</td>
<td>352</td>
<td>352</td>
</tr>
</tbody>
</table>

The t statistics are shown in parentheses.
Table 2. Estimated Pollution Functions for Each Crop

<table>
<thead>
<tr>
<th>Crop</th>
<th>Intercept</th>
<th>( \gamma_i )</th>
<th>( \gamma_{i,1} )</th>
<th>( \gamma_{i,2} )</th>
<th>( \gamma_{i,3} )</th>
</tr>
</thead>
<tbody>
<tr>
<td>Corn</td>
<td>-221.84</td>
<td>10.42</td>
<td>5.50</td>
<td>25.11</td>
<td>7.04</td>
</tr>
<tr>
<td>Barley</td>
<td>-118.36</td>
<td>10.08</td>
<td>5.97</td>
<td>1.19</td>
<td>5.09</td>
</tr>
<tr>
<td>Wheat</td>
<td>-7.04</td>
<td>10.72</td>
<td>5.97</td>
<td>2.13</td>
<td>1.12</td>
</tr>
<tr>
<td>Sunflower</td>
<td>-1.19</td>
<td>5.09</td>
<td>5.97</td>
<td>2.13</td>
<td>1.12</td>
</tr>
<tr>
<td>Rice</td>
<td>-0.11</td>
<td>4.66</td>
<td>5.97</td>
<td>2.13</td>
<td>1.12</td>
</tr>
</tbody>
</table>

- The t statistics are shown in parentheses.

The model are \( z_{i,t} \), crop acreage (ha), \( u_{i,t} \), nitrogen uptake by crop (kg/ha), and \( v_{i,t} \), nitrogen volatilization (kg/ha). Nitrogen uptake and volatilization are a proportion of yield \( (\gamma_i \cdot y_{i,t}) \) and fertilization \( (\delta_i \cdot n_{i,t}) \), respectively, where values for \( \gamma_i \) and \( \delta_i \) are taken from EPIC.

[32] The model parameters are \( p_i \), price of crop \( i \) (€/kg), \( p_w \), water price (€/m³), \( p_n \), active nitrogen price (€/kg), \( s_i \), crop subsidy payments (€/ha), \( k_i \), fixed costs (€/ha), \( \mu \), nitrogen cost damages (€/kg), \( \delta_i \), volatilization rate (%), \( \gamma_i \), nitrogen uptake rate (kg/t), \( r \), discount rate, and \( g_0 \), the initial value of \( g \). Parameter values are \( p_i \) from 90 to 240 €/t, \( p_w \) is 0.01 €/m³, \( p_n \) is 0.6 €/kg, \( s_i \) from 170 to 390 €/ha, \( k_i \) from 370 to 650 €/ha, \( \mu \) is 1.23 €/kg, \( \gamma_i \) from 2.5 to 28 kg/t, \( \delta_i \) is 0.024, \( r \) is 0.03, \( g_0 \) is 103 kg/ha. The parameter \( \mu \) is the unit emission cost of nitrogen leaching, and its value is calculated from the 22.6 mg N-NO₃/L level of nitrogen concentration in leachate (see percolation and nitrogen leaching in Table 5), and the 2.8 cents €/m³ cost of removing nitrogen from water at this level of concentration. The emission cost \( \mu \) is equal to 1.23 €/kg, because 44.25 m³ of water have to be treated to remove 1 kg of nitrogen. The 2.8 cents €/m³ cost of removing nitrogen is an engineering estimate of a discontinuous tertiary biological denitrification treatment.

[33] Constraints (5) and (6) are the crop succession and frequency equations on crop acreage \( z_{i,t} \), where \( c_i \) is the number of periods that crop \( i \) is planted and \( d_i \) is the number of interval periods without planting. Constraint (7) includes the Common Agricultural Policy set aside requirements \( (l = 10\%) \), and constraint (8) is the labor availability restriction where \( m_i \) is labor input by crop \( i \) and \( M \) is total labor available. Crops in the district are not labor intensive, and labor per hectare for each crop is considered constant.

[34] The soil nitrogen reserve \( q_{t+1} \) at the beginning of period \( t + 1 \) is given by equation (9) as an average of the soil nitrogen content of the acreage corresponding to each crop. Then \( g_{c,t+1} \) is set equal to \( q_{t+1} \), and \( g_{r,t+1} \) is introduced in the production function of each crop for period \( t + 1 \). The optimal control problem includes irrigation water and nitrogen fertilizer as control variables, and nitrogen stock in the soil as state variable, which is driven by the equation of motion, an identity linking inputs and outputs of nitrogen in the soil.

4. Results

[35] The study area is the Flumen-Monegros irrigation district in the Ebro basin of Spain, with 60,000 ha cultivated and 45,800 ha under irrigation. The climate is semiarid with insufficient rain throughout the entire year, making irrigation essential for profitable agricultural production. The origin of water resources is the Gallego river, through the Sotonera dam with 187 hm³ of capacity, and the Cinca river, through the Cinca canal system. Surface irrigation is the common irrigation technology, although sprinkle irrigation is being introduced in some locations. Maps of the spatial distribution of soil types and crops are useful, because crop yields, water irrigation, nitrogen fertilization, percolation and nitrogen leaching are linked to soil type. The main crops in the region are winter cereals (barley and wheat), corn, sunflower, rice and alfalfa, with statistical data available on crop acreage [Gobierno de Aragón, 1999]. The physical and socioeconomic characteristics of the irrigation district are described in the work of Martinez [2002], which
Martínez and Albiac

under Chacilla, Planteros, and Corraletes soils are given by

Results comparing pollution control instruments for corn,

functions for the six crops, and choosing Corraletes as the representative soil for the whole irrigation area.

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is 24.1 million C

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in the soil. The optimal solution differs from actual behavior

of farmers, which are using excessive quantities of nitrogen

and water to cover the risk of underapplying, as noted by

Orius et al. [2000]. The steady state solution under the baseline scenario shows that corn and rice are the more profitable crops, with an intensive use of inputs and high nitrogen emissions. Total quasi-rent for the district is 24.1 million €, using 191 hm³ of irrigation water and 4525 t of nitrogen fertilizer, and generating 1459 t of nitrogen pollution emissions (Table 4).

The first scenario simulates taxes on water which increase water prices from 1 cent to 6 and 9 cents €/m³. The reason for selecting the scenario of water prices equal to 6 and 9 cents €/m³ is that these are the prices being consid-

nitrate concentration on the Flumen River at Sarriñena, located at the end of the irrigation district. The time series cover the 1982–2003 period, with one day nitrate concentration measurements in January and September of each year. Nitrate concentration moves in the range 15 to 30 mg NO₃⁻/L, with a few extreme values reaching 35 and 10 mg/L in some periods.

The dynamic model is used to analyze the effects of different abatement measures to control nitrate pollution emissions, and to rank their cost efficiency. The measures examined are higher irrigation water prices, nitrogen fertilizer taxes, standards limiting nitrogen fertilization, and taxes on pollution emissions. Because no analytical solutions can be obtained, the model is solved numerically under these alternative scenarios, generating the time paths for variables yield, input use, nitrogen stock in soil and nitrogen leaching.

First, the steady state solution is generated under the base scenario of current crop production conditions in the district, and then the steady state solutions for the different alternative control measures are generated. Ranking of the pollution control measures is based on these steady state solutions, comparing welfare, quasi-rent and nitrogen leaching for the whole irrigation district. For input and emission tax instruments, welfare is equal to quasi-rent, minus pollution damage, plus taxes. This is so because quasi-rent calculations include taxes, so welfare is calculated with taxes added.

The baseline scenario approximates present conditions in the district in relation to yields, stock of nitrogen in the soil, crop and input prices, revenues and costs. It is assumed that farmers maximize quasi-rent, and are aware of the effects of production decisions on the nitrogen stock in the soil. The optimal solution differs from actual behavior of farmers, which are using excessive quantities of nitrogen and water to cover the risk of underapplying, as noted by Orius et al. [2000]. The steady state solution under the baseline scenario shows that corn and rice are the more profitable crops, with an intensive use of inputs and high nitrogen emissions. Total quasi-rent for the district is 24.1 million €, using 191 hm³ of irrigation water and 4525 t of nitrogen fertilizer, and generating 1459 t of nitrogen pollution emissions (Table 4).

The first scenario simulates taxes on water which increase water prices from 1 cent to 6 and 9 cents €/m³. The reason for selecting the scenario of water prices equal to 6 and 9 cents €/m³ is that these are the prices being consid-

includes details of the data on weather, costs and management operations by crop, and irrigation practices by the irrigation district associations.

Figure 1 presents the spatial distribution of the five agricultural soil types defined in the district based on the work by Nogue’s [2002], with the parameters shown in Table 3. All irrigated crops except rice, are planted mainly in Chacilla, Planteros, and Corraletes soils, while rice is planted only in Valfonda, which is the most salt-affected soil. An optimal control problem with 48 nonlinear production and pollution functions (six crops x four soils), is too complex to be solved even numerically. The problem has therefore been simplified by working with production and pollution functions for the six crops, and choosing Corraletes as the representative soil for the whole irrigation area. Results comparing pollution control instruments for corn, under Chacilla, Planteros, and Corraletes soils are given by Martínez and Albiac [2004], and results comparing the instruments for the six crops under surface and sprinkle irrigation technologies in a static setup are presented in the work of Martínez et al. [2002].

Although there is not specific data on nitrate concentrations in the irrigation district, the RICA network (Water Quality Integrated Network) provides ambient pollution data on some adjacent locations, which are related to the pollution emission levels (Confederación Hidrográfica del Ebro, Water Quality Data Series, available at http://www.oph.chebro.es/DOCUMENTACION/Calidad/pa3_6.htm). The RICA network provides information about

<table>
<thead>
<tr>
<th>Soil</th>
<th>Acreage, ha</th>
<th>WHCb Efficiencyc ECd</th>
</tr>
</thead>
<tbody>
<tr>
<td>AG0 Chacilla high</td>
<td>2,572</td>
<td>1,800</td>
</tr>
<tr>
<td>AG1 Planteros low</td>
<td>18,628</td>
<td>730</td>
</tr>
<tr>
<td>AG2 Corraletes intermediate</td>
<td>30,550</td>
<td>2,043</td>
</tr>
<tr>
<td>AG3 Valfonda low (high salinity)</td>
<td>7,942</td>
<td>2,680</td>
</tr>
<tr>
<td>AG4 - low (dryland)</td>
<td>2,229</td>
<td>1,350</td>
</tr>
</tbody>
</table>

*aBased on work by Nogue’s [2002]. The names of the reference series have been used in the text to designate agricultural units across the whole irrigation area.
bWater holding capacity (m³/ha).
cIrrigation efficiency (%).
dSalinity (electrical conductivity, dS/m).
Table 4. Results of Key Variables Under the Baseline Scenario by Crop

<table>
<thead>
<tr>
<th>Crop</th>
<th>Production, t/ha</th>
<th>Water Use, m³/ha</th>
<th>Nitrogen Use, kg/ha</th>
<th>Nitrogen Leaching, kg/ha</th>
<th>Quasi-rent, €/ha</th>
</tr>
</thead>
<tbody>
<tr>
<td>Corn</td>
<td>14.1</td>
<td>6.220</td>
<td>325</td>
<td>140</td>
<td>1,180</td>
</tr>
<tr>
<td>Barley</td>
<td>6.0</td>
<td>2.200</td>
<td>180</td>
<td>29</td>
<td>375</td>
</tr>
<tr>
<td>Wheat</td>
<td>6.6</td>
<td>3.500</td>
<td>140</td>
<td>32</td>
<td>550</td>
</tr>
<tr>
<td>Sunflower</td>
<td>2.9</td>
<td>3.100</td>
<td>70</td>
<td>20</td>
<td>470</td>
</tr>
<tr>
<td>Alfalfa</td>
<td>17.3</td>
<td>7.800</td>
<td>70</td>
<td>15</td>
<td>740</td>
</tr>
<tr>
<td>Rice</td>
<td>5.6</td>
<td>12.000</td>
<td>170</td>
<td>57</td>
<td>797</td>
</tr>
</tbody>
</table>

Table 5. Results of Alternative Policy Measures in the District

<table>
<thead>
<tr>
<th>Measure</th>
<th>Welfare, 10⁶ €</th>
<th>Quasi-rent, 10⁶ €</th>
<th>Water, hm³</th>
<th>Nitrogen, t</th>
<th>Percolation, hm³</th>
<th>Nitrogen Leaching, t</th>
</tr>
</thead>
<tbody>
<tr>
<td>Base Scenario</td>
<td>22.3</td>
<td>24.1</td>
<td>190.7</td>
<td>4,525</td>
<td>66.1</td>
<td>1,459</td>
</tr>
<tr>
<td>Water price, 0.06 €/m³</td>
<td>21.2</td>
<td>18.8</td>
<td>86.4</td>
<td>4,367</td>
<td>43.3</td>
<td>1,381</td>
</tr>
<tr>
<td>Water price, 0.09 €/m³</td>
<td>19.6</td>
<td>12.6</td>
<td>109.1</td>
<td>4,039</td>
<td>20.2</td>
<td>1,346</td>
</tr>
<tr>
<td>Nitrogen price, 0.90 €/kg</td>
<td>22.4</td>
<td>22.6</td>
<td>200.6</td>
<td>4,265</td>
<td>45.3</td>
<td>1,222</td>
</tr>
<tr>
<td>Nitrogen price, 1.20 €/kg</td>
<td>22.7</td>
<td>21.5</td>
<td>186.6</td>
<td>3,976</td>
<td>56.2</td>
<td>990</td>
</tr>
<tr>
<td>Nitrogen standard</td>
<td>23.7</td>
<td>23.8</td>
<td>98.1</td>
<td>4,134</td>
<td>14.1</td>
<td>634</td>
</tr>
<tr>
<td>Emission tax</td>
<td>23.9</td>
<td>23.8</td>
<td>185.4</td>
<td>3,596</td>
<td>43.4</td>
<td>697</td>
</tr>
</tbody>
</table>
on nitrogen fertilizer increase welfare slightly, between 0.1 and 0.4 million euros, while achieving a moderate abatement. Higher water prices are very inefficient since welfare falls between 1.1 and 2.7 million euros, with an almost negligible leaching abatement of less than 8%. This result questions the reliance on higher water prices of the recently approved Water Framework Directive of the European Union, because the water pricing instrument seems relatively inefficient to abate nonpoint pollution and contribute toward reaching the “good status” target for all waters.

Another interesting outcome of the study is that the effects of the above nitrogen pollution abatement measures can be compared with the impact on nitrogen pollution of the domestic National Irrigation Plan legislation, which subsidizes investments to modernize irrigation technologies. Uku [2003] has examined the nitrogen pollution impact of the National Irrigation Plan in the Cinco Villas county irrigation area, which covers 55,400 ha and is located 70 km west of the Flumen district. In Cinco Villas, 5% of the cultivated acreage is taken up with tomatoes and peppers, in addition to the crops cultivated in Flumen. The Cinco Villas results indicate that nitrogen standards reduce emissions by almost 40%, at a cost of a 10% reduction in quasi-rent. Modernizing irrigation technologies, without including the investment costs of secondary canals and plot irrigation systems, increases quasi-rent by almost 30% and reduces emissions by more than 70%. However, the necessary investments cannot be financially sustained by the additional income from higher crop yields, even when public subsidies from the plan are accounted for. The only crops that can sustain financially the investment, with or without subsidies, are tomatoes and peppers. However, these high-profit crops are highly input and pollution intensive, and the implication is that the National Irrigation Plan may induce the expansion of high-profit crops, which would further increase pollution and degrade the quality of water resources. Consequently, the abatement measures examined above appear preferable to the National Irrigation Plan legislation, aimed at modernizing irrigation technologies.

In line with the results presented here, several studies from the nonpoint pollution literature confirm that the first best measure of taxing nitrogen emissions is the social optimum measure [Pan and Hodge, 1994; Johnson et al., 1991]. Among the second best measures, our results indicate that a standard on nitrogen fertilization is the preferred measure in terms of cost efficiency, a finding that is supported by several studies on nitrate pollution [Mapp et al. [1994] in USA, Schmid [2001] in Austria, Ribaudo et al. [2001] for basin-wide pollution reductions below 26% in USA, and Martinez et al. [2002] in Spain]. However, it should be pointed out that the abatement costs estimated in reducing nitrogen leaching in the studies by Mapp et al., Schmid, and Martinez et al. are somewhat higher than the costs calculated here.

The introduction of pollution markets involving the use of nitrogen emission permits has been simulated in the literature by Thomas and Boisvert [1994] and Kampas and White [2003], and this permit instrument can be compared with the emission tax instrument examined here. The costs of abatement permits estimated by Thomas and Boisvert are higher than our estimated costs from emission taxes, in terms of the relative reduction in quasi-rent and pollution. No direct comparison is made with the Kampas and White results because their abatement costs are contingent upon the particular permit allocation rule implemented by the regulator.

A further contribution from the study is that our results may shed some light on the discussion about the appropriate instrument base for pollution control, which depends upon the correct understanding of the key underlying biophysical processes. The issue raised by our empirical findings is the following: instruments based on nitrogen use seem more cost-efficient than instruments based on irrigation water, in contradiction to the results obtained by Helfand and House [1995] and Larson et al. [1996], which are reported by Horan and Shortle [2001].

Since ranking of policy instruments is highly dependent on characteristics linked to particular physical, climatic, and agricultural conditions, both nitrogen use and irrigation water may be appropriate base instruments, and no sweeping or general recommendations should be advanced from results at individual sites. However, if the dynamic aspects really are key features in nitrogen biophysical processes, then neglect of dynamics may lead to incorrect choices not only of instrument levels but also of the instrument base.

Helfand and House [1995] and Larson et al. [1996] examine the cost efficiency of control measures for lettuce production under two heterogeneous soils in California’s Salinas Valley, in order to abate nitrogen leaching by 20%. Helfand and House indicate that a nitrogen tax and a nitrogen standard are both very inefficient measures with very high welfare costs. This finding is contrary to our empirical evidence, which indicates that the second best preferred measure is a nitrogen standard. Our results indicate that a 9% reduction of nitrogen fertilizer reduces leaching by half, and that taxing nitrogen fertilizer is much more cost efficient than taxing irrigation water. Furthermore, nitrogen pollution results from Martinez and Albiac [2004] analyzing corn production on three soil classes, in which soil nitrogen dynamics are included, indicate that reducing corn fertilization by 20% cuts nitrogen leaching by more than half.

In contrast, Helfand and House [1995], working with a static model of lettuce production on two soils, find that costs of a nitrogen standard or tax to be prohibitive, a 90% reduction in nitrogen fertilization being required to achieve a 20% decrease in leaching within the same production period. The problem with their water irrigation standard or tax instruments is that a reduction in water irrigation will curb percolation and nitrogen leaching in the current production period, but since nitrogen application is unabated, nitrogen content will build up in the soil, and nitrogen leaching in subsequent periods will increase through irrigation or rainwater events. The key question therefore is whether or not the dynamic behavior of nitrogen in the soil is important.

The claim that nitrogen soil dynamics is the factor making nitrogen use a more appropriate base instrument than water use, has been tested by using an alternative model specification, where pollution damages are immediate. When the equation of motion (equation (4)) representing the dynamics of nitrogen in the soil is eliminated from the model, crop yields, nitrogen stock in the soil, and nitrogen leaching become static and constant for all periods.
Results from the modified static model show that quasi-rent losses from each nitrogen abatement measure are close to losses under the dynamic model, but abatement effects on nitrogen leaching are now inverted with respect to the instrument base. Instruments based on nitrogen use are now quite substantially less efficient in reducing nitrogen pollution than instruments based on water irrigation, with abatement through nitrogen-based instruments in the district falling from between 20–50% to less than 10% of the nitrogen leaching baseline. This finding is confirmed in the Flumen irrigation district for the case of corn production nitrogen leaching baseline. The implication is that since key intertemporal biophysical processes, implicit in soil nitrogen dynamics, are not included by Helfand and House [1995], their cost-efficiency results of nitrogen standards and taxes are questionable, and could be biased.

A final important issue on nonpoint pollution control is the possibility of discriminating the application of measures by individual site characteristics such as crop or soil type. The results presented here and those of Martínez and Albiac [2004] on nitrate pollution from corn under three heterogeneous soils indicate the possibility of differentiating application of control measures by crop and soil type. Both studies demonstrate that the preferred second-best instrument is a standard, but application could take into account the pollution potential of each crop and the vulnerability of each soil type.

In the absence of abatement measures, wheat, barley, sunflower, and alfalfa all have leaching levels below 30 kg/ha (Table 4), so these crops would not be subject to abatement, while other more polluting crop production activities such as corn and rice would be subject to control. However, when standards are applied only to the more polluting crops, corn and rice, no welfare gains are detected comparing with standards applied to all crops.

Small welfare increases are found by Martínez and Albiac [2004] when applying differentiated nitrogen input standards to corn nitrogen leaching by soil type, a result similar to that of Helfand and House [1995] for lettuce in two soils. Corn nitrogen leaching in Chacilla soils is just 45 kg/ha, which is below corn leaching levels in other soil types under any abatement measure. However, when a standard is applied only in Planteros and Corraletes soils, there is a small welfare gain over applying the standard to all soils. The implementation of a nitrogen standard in Planteros and Corraletes soils but not in Chacilla, improves welfare by 1% over a homogeneous nitrogen standard for corn in the district.

Discriminating control measures could depend on the particular combination of crop planted and soil chosen; thus linking control to the land use decision by the farmer. In the Flumen-Monegros irrigation district case, wheat, barley, sunflower, and alfalfa would not be subject to pollution abatement instruments, while more polluting crop production activities, such as corn and rice cultivation would be subject to control only in vulnerable Planteros and Corraletes soils. This discrimination by crop and soil could be implemented by applying a land use tax, adapted to crop and soil, provided that the land tax were adequate in terms of cost efficiency and transaction costs. The empirical evaluation of this land use tax will be facilitated by a modeling effort that captures nitrogen soil dynamics for each crop and soil class.

Conclusions

Evaluation of the efficiency of alternative nitrogen abatement measures requires the consideration of biophysical aspects linked to the dynamics of nitrogen in the soil, taking into account crop type and soil class. The effects of selected abatement measures have been examined through a dynamic model, which includes six crops and one representative soil, in the Flumen-Monegros irrigation district located in the Ebro basin of Spain. Ranking the nitrogen control instruments by their cost efficiency contributes to the information needed in the policy decision process. The results obtained agree with previous literature, and indicate that a fertilizer standard is the more efficient second-best measure to control nitrogen pollution.

Several measures to reduce emissions have been simulated and compared to the present baseline scenario. An increase in water prices only slightly reduces nitrogen discharges at very high costs to farmers and society. A tax on nitrogen fertilization results in more significant pollution reduction at much lower costs. A standard on nitrogen application curbs emissions by more than half, with a very moderate impact on quasi-rent and gains in welfare. The introduction of subsidies linked to the standard could be a good second-best instrument to achieve nitrogen pollution control.

The finding that higher water prices are very inefficient to abate emissions, questions the reliance of the European Water Framework Directive on water pricing as a pollution instrument to reach the “good status” target for all waters. The implication is that other instruments included in the directive, such as ambient quality standards and emissions limits, need to be applied in order to curb pollution. Looking at the Spanish domestic policies, the main piece of legislation affecting nonpoint pollution is the National Irrigation Plan, which promotes irrigation modernization through public subsidies. Although yields increase and pollution is reduced substantially by renovating secondary canals and plot irrigation systems, the problem is that the required investments are not financially sustainable, even when public subsidies are accounted for. The consequence is that nitrogen pollution could be controlled by the abatement measures examined here, but not by the National Irrigation Plan legislation.

Additionally, the results obtained contribute with further evidence to the discussion on the choice of the appropriate instrument base for nitrogen control. Horan and Shortle [2001] state that instruments based on irrigation water are more cost-efficient than instruments based on nitrogen fertilization, using the empirical results by Helfand and House [1995] and Larson et al. [1996]. The reason given is that irrigation water is more highly correlated with nitrate leaching, implying that the appropriate instrument base is not the nutrient responsible for pollution but rather the input most highly correlated with pollution. This interpretation appears inaccurate, because the dynamics of nitrogen in the soil are ignored, and in fact, by including...
and excluding soil dynamics in the model, it is shown that nitrogen based instruments become more efficient when soil dynamics are accounted for. This discussion proves that nonpoint pollution control instruments cannot be assessed accurately without a correct understanding of key underlying biophysical processes, which need to be incorporated in order to evaluate control instruments. Neglect of the dynamic aspects of nonpoint pollution may have serious consequences for the design of policy measures.

A quantitative assessment has also been performed, to explore the option of using discriminating abatement measures, and implementing control only for crops with high pollution potential on vulnerable soils. Differentiating the application of control by crop and soil class may enhance welfare, although the size of this welfare gain needs to be evaluated in the general case of several crops under different soil types. No welfare gains are found from differentiated control in the case of several crops in a single representative soil, and only small welfare gains are documented for the case of differentiated control of corn on three soils. However, these results are contingent upon the level of damage costs from nitrogen pollution, because in areas with valuable ecosystems and strong nitrogen damage costs, welfare gains from differentiated control would be higher and may justify discriminating control measures.

An important question for the choice of the correct pollution control instrument, is the implementation costs of the instruments. Measures that seem suitable may be associated with implementation difficulties relating to their political acceptability or transaction costs, and policy makers should evaluate the trade-off between cost efficiency and simplicity of implementation.

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Figure 1. Map of soil types in the Flumen-Monegos irrigation district (based on the work by Nogués et al. [1999] and Nogués [2002]).