Agroeconomic Analysis of Nitrate Crop Source Reductions

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Abstract: This paper presents an agroeconomic approach to assess the economic impact of improving nitrogen and irrigation management practices in California's Tulare Lake Basin and the Salinas Valley. The approach employs a self-calibrated mathematical programming model with a constant elasticity of substitution production function and two nests: one for irrigation and one for nitrogen. Agricultural crop yields are maintained as a worst-case for improving nitrogen use efficiency. Small reductions (<25%) in nitrate load to groundwater can be achieved at relatively low costs. Load reductions of 50% may require more costly nitrogen management practices and a broader education strategy with higher reductions in farm net revenues and irrigated area. Other policy instruments such as a tax and levees on applied nitrogen may help reduce groundwater load and raise revenues for alternate drinking water supplies in affected areas. The model also provides further evidence that it is possible to integrate agronomic and economic models that account for substitutability of applied nitrogen and water in agricultural production for policy analysis. **DOI: 10.1061/(ASCE)WR.1943-5452.0000268.**

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Introduction

Improving nitrogen and water management on croplands is important for reducing nitrate groundwater contamination. Nitrogen, soil, and water management practices can reduce agricultural effects on groundwater quality (Harter et al. 2012). However, new practices often require increasing management intensity, which changes costs and profitability of farming. This work develops a novel method of estimating the economic impacts of policies that reduce nitrogen loading to groundwater from crop-farming activities. California's Tulare Lake Basin and the Salinas Valley as used as case studies. These agricultural areas have high agricultural crop and dairy production value; however, these regions also have a significant proportion of population vulnerable to groundwater nitrate contamination of drinking water.

Widespread application of synthetic nitrogen fertilizers is a foundation for California's robust agricultural economy. However, excessive use has contaminated groundwater throughout California's agricultural regions (Burrow 2010; Zhang et al. 1998). Nitrate in groundwater is a public health concern. Many Californians rely on groundwater as their primary drinking water source [Department of Water Resources (DWR) 2003], and ingesting excessive nitrate is linked to several health problems (Ward et al. 2005). Agriculture is both the largest contributor of nitrate to groundwater and a primary driver of local economies in the Tulare Lake Basin and the Salinas Valley, as the five counties in these two regions are among the nation's most agriculturally productive.

Various technologies and practices can help farmers use nitrogen more effectively and reduce nitrate leaching. Conventional wisdom suggests that a reduction of nitrate loading will increase management and production costs, reducing profit. The dual goals of maintaining profitability and reducing leaching may not always be at odds, and nitrogen monitoring may be a possible low-cost or even profitable strategy (Hartz 1994; Knapp and Schwabe 2008). Identifying and implementing practices that attain these dual goals can help preserve the rural agricultural economy and groundwater quality.

In practice, farming operations often change several practices simultaneously. Suites of practices can increase nitrogen use efficiency and decrease pollution potential (Broadbent and Carlton 1978; Letey et al. 1982; Meyer and Marcum 1998; Stark et al. 1983). Combinations of production practices can be thought of

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as "bundles" of practices as they coproduce the desired benefits. As with individual practices that decrease leaching, bundles typically require capital costs for technology and additional operational costs by moving towards more intensive and expensive labor use. Few studies quantify the costs of implementing technology bundles or their impacts on nitrogen loading. Knapp and Schwabe (2008) offer an example of a dynamic multiyear approach that accounts for water and nitrogen application as well as irrigation system bundles. This present research accounts for water and nitrogen use efficiency while focusing on the economics of nitrate leaching to groundwater under different regulatory and economic-based policy scenarios.

Modeling the interaction between nitrogen fertilizer, irrigation, crop mix, crop yield, and the costs and revenues of agricultural production is complex and involves several uncertainties. Previous research has focused on the policy aspects of regulating nitrates, with less attention to economics. Daberkow et al. (2008) review literature on economic modeling of public policies for changing nitrogen use practices in agriculture. In general, farm income falls from taxes on fertilizer or nitrogen effluent, or setting more stringent limits on nitrogen application or effluent discharge. Effectiveness and costs vary across studies, but the literature seems to concur that modest improvements in nitrogen use efficiency may have little cost to farm net income (Knapp and Schwabe 2008).

Many policies to help reduce groundwater nitrate loading vary in effectiveness and ease of application (Canada et al. 2012). Variability and heterogeneity in production affect the effectiveness and equity of any policy. Individual taxes based on the emissions (or nitrate leaching in this case) could be applied to attain a socially optimal solution, but such taxes can be costly to apply (Canada et al. 2012). Helfand and House (1995) evaluated second-best policies including uniform taxes, uniform rollbacks, single taxes on nitrogen use water, and prescriptive reductions in nitrogen or water use. Second-best policy instruments, such as output taxes, uniform taxes, or cutbacks, may be close to the best performing policy and are often easier to apply. They also found that taxing applied nitrogen alone can be more costly than taxing water alone (Knapp and Schwabe 2008). Johnson et al. (1991) modeled a 25% reduction in applied nitrogen, restrictions on nitrate leaching, a tax on applied nitrogen, and a tax on effluent and found that small reductions can be achieved by noncostly practices, but larger reductions come at higher costs. Wu et al. ~~(1993)~~ simulated choice of irrigation investment and crop in response to effluent taxes, input taxes, and restrictions in applied water over a 10-year period considering soil conditions. In their case study, for a three-crop system in a small agricultural region in Oklahoma, they found that a tax on nitrogen alone performed poorly compared with other alternatives.

The current approach models basin-scale long-term costs to agriculture from restricting nitrate load to groundwater, applying a tax on applied nitrogen, or applying a penalty for a nitrate load in excess of a given threshold. The lump sum of these taxes is not assumed to return to the industry. Unlike previous work, this work is concerned with nitrogen use efficiency (NUE) expressed as partial nutrient balance (PNB), water use efficiency, and their respective tradeoffs with respect to investments in NUE and irrigation efficiency improvements. The approach follows a long-term mass balance approach that links PNB to irrigation efficiency. A sensitivity analysis examines increases in the marginal costs of improving nitrogen use efficiency.

Case Studies: The Tulare Lake Basin and the Salinas Valley in California

To quantify the economic cost of nitrogen use efficiency in California, the Tulare Lake Basin (TLB) and the Salinas Valley (SV) in California are used as case studies. The Tulare Lake Basin includes four counties in California's southern Central Valley, which encompass about one-third of the state's irrigated crop area (DWR 2009) and total crop revenues [Agricultural Issues Center (AIC) 2009]. More than 200,000 t of nitrogen are applied to crops each year in this area.. More than 50% of all California's dairy production value is located in the study area, although surplus nitrogen applications from manure are not addressed or considered in this study. Irrigation water is from groundwater (33%), federal and state water project imports (37%), and local surface water sources (30%) (DWR 2009).The Salinas Valley is located on the central coast of California, about 100 km west of the TLB. This region has high-value specialty crops including berries, vine crops, and vegetables, many of which are unique in the United States. In the SV, irrigation with groundwater is predominant, and higher efficiency irrigation methods are more common than in the TLB. However, some vegetable and berry crops pose a higher risk of nitrogen leaching into groundwater, because less of the applied nitrogen is removed by harvest.

The TLB and the SV contain rural communities and some urban centers deemed as vulnerable to drinking water nitrate contamination (Harter et al. 2012). This research estimates economic costs of reducing nitrate load to groundwater in these areas from crop farming, a major source of groundwater nitrate. The wide variety of crops the Tulare Lake Basin and Salinas Valley cover 1.44 million and 92,000 ha, respectively. These include alfalfa, almonds and pistachios, corn, cotton, grain and field crops, lettuce, orchards, strawberries, subtropical, tomato, vegetables, and vine crops. Full details on the crop share for each region are shown in Dzurella et al. (2012).

Methods

A self-calibrated profit-maximizing model of agricultural production is developed to assess the economic impact on farmers attributable to policies that reduce nitrogen loading from croplands. Because nitrogen loading to groundwater in irrigated cropping systems is largely a function of nutrient and water management, the model is based on economic and environmental consequences of changes in nutrient use and irrigation efficiency. Here, better management requires additional monetary inputs (e.g., for infrastructure labor and information and education to reduce nitrogen loading from croplands). The model allows for tradeoffs between monetary investments in production inputs (management practice bundles) and total nitrogen and water use. The model maximizes profits from farming while constraining yields to be constant.

Conceptual Model Framework: Partial Nitrogen Balance, Nitrogen Surplus, and Irrigation Efficiency

Nitrate leaching from irrigated croplands to groundwater is considered to be a function of the long-term (multiannual) mass balance between total nitrogen applied to cropland and nitrogen removed by harvest, atmospheric losses, and runoff (net long-term changes in landscape nitrogen storage are assumed negligible). The nitrogen mass balance is effectively controlled by water application (quantity and timing) relative to crop water use and by nitrogen management (quantity and timing) relative to crop nitrogen needs. This modeling accounts for both water use efficiency and nitrogen use efficiency improvements affecting nitrate leaching.

As a measure of nitrogen use efficiency, this current model is based on two interrelated metrics that, together, represent nitrate leaching potential: partial nutrient balance and nitrogen surplus. Partial nutrient balance is the ratio of the total nitrogen removed by the crop, $\tilde{N}$, to nitrogen applied, N. The nitrogen removed is

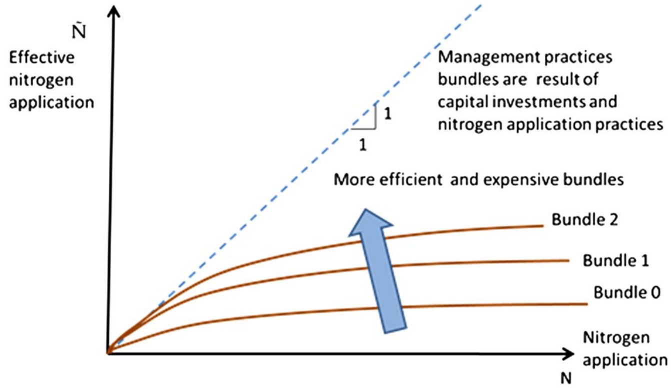

Fig. 1. Effective nitrogen versus applied nitrogen by management practice bundle; bundle 0 refers to practices before any improvements; bundles 1 and 2 refer to more efficient and expensive bundles

also called the *effective* amount of nitrogen, $\tilde{\mathrm{N}}$, which is generally smaller than the nitrogen applied, N. The complement of PNB, 1-PNB, is a common measure for nitrogen surplus remaining in the field after accounting for harvest removal. The actual nitrogen surplus, the difference between nitrogen applied and that taken up by the crop, is N multiplied by (1-PNB). It is subject to groundwater leaching, surface runoff, and atmospheric losses. If the total applied nitrogen equaled the effective nitrogen (PNB $= 1$) at any level of nitrogen application, N, the nitrogen efficiency curve, $\tilde{\mathrm{N}}(\mathrm{N})$, would yield a straight line with a 1:1 slope (Fig. 1).

Management practice bundles requiring specific capital and other investments are represented in terms of their nitrogen use efficiency curves $\tilde{\mathrm{N}}(\mathrm{N})$. For each practice bundle, nitrogen use efficiency at low N application rates tends to be very high (albeit with low yields), and the value of $\tilde{\mathrm{N}}(\mathrm{N})$ is close to the 1:1 line. As the N application rate increases, nitrogen uptake into harvest typically decreases relative to the amount of nitrogen applied. Hence, the $\tilde{\mathrm{N}}(\mathrm{N})$ curve levels off relative to the 1:1 line of $\tilde{\mathrm{N}}(\mathrm{N})$ (Fig. 1). Plotting such curves for various (hypothetical) management practice bundles on a single graph allows for the comparison of the nitrogen use efficiency (expressed as PNB) of various practices. Bundles with lower slopes have smaller PNB and are less desirable (e.g., bundle 0), whereas bundles with steeper slopes (i.e., higher PNB) are preferred.

This work uses a substitution relationship between capital investments for efficient nitrogen use and total nitrogen use calibrated to surveyed costs of application bundles. These tradeoff curves follow a constant elasticity of substitution (CES) functional form and assume effective nitrogen use remains constant. One challenge in this approach is that bundles at the farm level are discrete costs, i.e., they are either adopted or not adopted by the farmer, and therefore must be approximated to a nonlinear function as shown in Dzurella et al. (2012). The maximum entropy approach employed to estimate the CES relationship allows estimation of expected values of parameters with small or incomplete data sets (Shannon 1948; Paris and Howitt 1998).

Likewise for irrigation efficiency, it is assumed that bundles with higher irrigation efficiency require capital investments to maintain crop yields. Irrigation efficiency is measured as ratio of ET over applied water. Hatchett (1997) parameterized this relationship for the Central Valley.

Information on irrigation technology and an approximation of the tradeoffs between capital investment and efficiency exists from previous studies (Hatchett 1997). However, with the exception of Knapp and Schwabe (2008), few analyses have compared the cost of improved nitrogen management practices, crop PNB (or other NUE measures), and the economics of nitrogen leaching to groundwater. The following section presents a model formulation and assumptions to derive such relationships for nitrogen management bundles.

Model Formulation

This model follows a multistep calibration process using a CES function with two nests: effective water and effective nitrogen. In the first step, a Leontief technology is employed that allows no substitution among inputs. The production function for the farmer for each crop includes six variable inputs: land, water, supplies, applied nitrogen, capital investments in nitrogen use efficiency, and capital investments in water use efficiency.

The variable supplies aggregates the costs of miscellaneous variable inputs, including labor and farming supplies other than nitrogen and water, which have been lumped into an amalgam of variable production costs per acre. In this program, *capital investments* are expenditures on equipment, management, and operation costs, which may include additional training of personnel, increased supervision, and crop consulting services. Two tradeoff curves exist in the model: one for water versus water capital investments and another for nitrogen and nitrogen capital investments. Medellín-Azuara et al. (2012a) present the full set of equations of a similarly nested model for irrigation efficiency only. In the present application, a simplified set of equations for multistep calibration is provided, with additional details in the Notation section of this paper. In the first step, the objective function [Eq. (1)] is given by

$$\max Z = \sum_{g}\sum_{i}\left(V_{gi}Y_{gi}\mathrm{XL}_{gi} - \sum_{j}a_{gij}\mathrm{XL}_{gi}\omega_{gij}\right) \quad (1)$$

where Z = net returns to land and management; $\mathrm{XL}_{gi,\mathrm{land}}$ = decision variable (land allocated for each crop i in each region g); and V_{gi}, Y_{gi} = prices and yields, respectively, for crop i in region g. On the cost side, the parameters a_{gij} and ω_{gi} are, respectively, the Leontief production and the unit cost coefficients for production inputs.

The program is constrained in Eq. (2) to a limiting amount of water and land:

$$\sum_{g}a_{gij}\mathrm{XL}_{gij} \le b_{gj}\, j \in \{\mathrm{land}, \mathrm{water}\} \quad (2)$$

Three other inputs include effective water, effective nitrogen, and supplies, where

$$a_{gi,\mathrm{EffW}}\mathrm{XL}_{gi} = \mathrm{ETAW}_{gi} \quad \forall\ g, i \quad (3)$$

$$a_{gi,\mathrm{EffN}}\mathrm{XL}_{gi} = \mathrm{AppN}_{gi}\mathrm{PNB}_{gi} \quad \forall\ g, i \quad (4)$$

$$a_{gi,\mathrm{Supl}}\mathrm{XL}_{gi} = \mathrm{SUPPL}_{gi} \quad \forall\ g, i \quad (5)$$

In Eq. (3), the left-hand-side or *effective water* is equal to the base estimated evapotranspiration of applied water for crop i in region g in volume units. Likewise, the *effective nitrogen* in Eq. (4) is defined as the proportion of the applied nitrogen taken by crop i in region g, in mass units. Finally, Eq. (5) assigns the cost of total supplies to crop i in region g, in monetary units.

The objective function [Eq. (1)] maximizes net returns to land and management for a limited amount of land, water, and for a given amount of supplies, water, and nitrogen use efficiency. By comparing the optimized values for land, water cost, nitrogen cost, and crop allocation (including costs of increasing efficiency) at different water and nitrogen use efficiencies, the cost of improving nitrogen use efficiency can be compared, which, in turn, could decrease groundwater pollution. The modules for nitrogen application efficiency versus capital investments in nitrogen use efficiency and the module of water capital investments versus irrigation efficiency are described subsequently.

Water Capital Investments versus Irrigation Efficiency

Capital investments on improved irrigation efficiency versus total applied water can be modeled following Hatchet (1997). The evapotranspiration of applied water for each crop is used as a proxy for irrigation efficiency:

$$a_{gi,\mathrm{EffW}}\mathrm{XL}_{gi} = \tau_{Wgi}\{[\beta 1_{gi} a_{gi}\mathrm{XL}_{gi}]^{\rho_{Wi}} + [(1-\beta 1_{gi}) a_{gi}\mathrm{XL}_{gi}]^{\rho_{Wi}}\}^{1/\rho_{Wi}} \tag{6}$$

Information to calibrate this component is taken from Hatchet (1997). The effective water amount on the left-hand side is as specified in Eq. (3). The parameters τ_{gi} and $\beta 1_{gi}$ are, respectively, the scale and the share factors in the CES functional form. On the right-hand side, XL_{gi} times a_{gij} for applied water and capital investments in applied water represent factors within the water efficiency nest that may substitute for one another. Finally, ρ_{Ni} is given by the elasticity of substitution σ_{Ni} of crop i, such that $\rho_{Ni} = (\sigma_{Ni} - 1)/\sigma_{Ni}$.

Investments and Costs for Increasing Nitrogen Use Efficiency

The second nest component [Eq. (7)] represents tradeoffs between nitrogen application and costs for improving nitrogen use efficiency, assuming agricultural yields are not reduced by these improvements. Again, a constant elasticity of substitution relationship is employed between the quantity of applied nitrogen and the costs of nitrogen application in Eq. (7), such that

$$\begin{aligned} a_{gi,\mathrm{EffN}}\mathrm{XL}_{gi} = \tau_{Ngi}\{&[\beta 2_{gi} a_{gi,\mathrm{AppN}}\mathrm{XL}_{gi}]^{\rho_{Ni}} \\ &+ [(1-\beta 2_{gi}) a_{gi,\mathrm{CPNB}}, \mathrm{XL}_{gi}]^{\rho_{Ni}}\}^{1/\rho_{Ni}} \end{aligned} \tag{7}$$

where the left-hand side (effective nitrogen) is as specified in Eq. (4) and corresponds to the vertical axis in Fig. 1. On the right-hand side, applied nitrogen and capital investments in PNB are the substitutable factors in this second nest. The rest of the parameters are as in the water efficiency nest [Eq. (6)].

In this case, the substitution parameter σ_{Ni} was estimated empirically using a maximum entropy approach, as only a small data set existed for PNB versus costs per unit area required for that particular PNB. Maximum entropy theory (Jaynes 1957; Shannon 1948; Paris and Howitt 1998) makes maximum use of the existing information to estimate a probability distribution for a particular parameter.

Finally, a calibration constraint on XL_g [Eq. (8)] restricts land to observed values $\tilde{X}_{gi}$, where ε is a small perturbation to decouple resources [Eq. (2)] and calibration constraints (Howitt 1995):

$$\mathrm{XL}_{gi} \leq \tilde{X}_{gi} + \varepsilon \quad \forall\ g, i \tag{8}$$

Once a solution to the linear program of these eight equations is found, the second step in this model uses the Lagrangian of the land use constraint to estimate a PMP quadratic cost function (Howitt 1995). In the third and fourth steps, the parameters for the CES water efficiency and nitrogen use efficiency are obtained. The resulting calibrated program is given by Eqs. (9)–(12):

$$\begin{aligned} \max\ \mathrm{NL2} = &\sum_g \sum_i v_{gi}\left[\tau_{2gi}\left(\sum_{j'} \beta_{gij'}\mathrm{XNN}_{gij'}\right)^{\rho_2}\right]^{1/\rho_2} \\ &- \sum_g \sum_i \sum_j (\delta_{gij}\mathrm{XNN}_{gij} + \gamma_{gij}\mathrm{XNN}_{gij}^2) \end{aligned} \tag{9}$$

where NL2 = net revenues for all regions and crops in the objective function. In this step, the decision variable is a vector of inputs XNN_{gi}. In this case, j', a subset of j, contains four elements: land, effective water (first nest), effective nitrogen (second nest), and supplies. These combine into the main CES production function in the first term in Eq. (9). The second and last term in Eq. (9) is the calibration quadratic PMP cost function (Howitt 1995). On the basis of the preceding objective function, two nested CES and a resource constraint are as follows:

$$\begin{aligned} \mathrm{XNN}_{gi,\mathrm{EffW}} = \tau_{Wgi}\{&[\beta 1_{gi}\mathrm{XNN}_{gi,\mathrm{water}}]^{\rho_{Wi}} \\ &+ [(1-\beta 1_{gi})\mathrm{XNN}_{gi,\mathrm{watercap}}]^{\rho_{Ni}}\}^{1/\rho_{Wi}} \end{aligned} \tag{10}$$

$$\begin{aligned} \mathrm{XNN}_{gi,\mathrm{EffN}} = \tau_{Ngi}\{&[\beta 2_{gi}\mathrm{XNN}_{gi,\mathrm{AppN}}]^{\rho_{Ni}} \\ &+ [(1-\beta 2_{gi})\mathrm{XNN}_{gi,\mathrm{CPNB}}]^{\rho_{Ni}}\}^{1/\rho_{Ni}} \end{aligned} \tag{11}$$

$$\sum_i \mathrm{XNN}_{gi} \leq b_{gi} \quad \forall\ g, j \in \text{land, water}\} \tag{12}$$

The mass balance and policy constraints are described next. Modifications to the mass balance constraints and costs of inputs [second term in Eq. (6)], allow for the modeling of the cost of different policy options.

Nitrogen Load to Groundwater from Agricultural Production

To estimate N load to groundwater in irrigated systems, a simplifying assumption is that PNB cannot exceed the irrigation efficiency, as irrigation water is the primary mobilizing flow for nitrogen to groundwater in these regions. In other words, farmers that employ efficient irrigation practices are more likely to adopt (or to already use) more efficient nitrogen application practices. In addition, to compute groundwater N loading, it is assumed that 10% of applied nitrogen is lost to the atmosphere as ammonia, nitrogen oxides, or dinitrogen gas. The remaining 90% of the (annually) applied N is either taken up by the crop or leached to groundwater (no significant runoff). The groundwater nitrogen load will therefore be between zero and the difference of PNB subtracted from 90%. The maximum potential fraction of nitrogen that can leach into groundwater is

$$\mathrm{GW}_{\mathrm{NO_3 load},gi} = \mathrm{Max}\{0, \mathrm{XNN}_{gi,\mathrm{AppN}}(0.9 - \mathrm{PNB}_{gi})\} \tag{13}$$

where $\mathrm{GW}_{\mathrm{NO_3 load},g,i}$ in Eq. (13) = groundwater nitrogen load; and the rest of the terms are as previously defined. The nitrogen load to groundwater is always nonnegative, thus the minimum value in Eq. (13) is zero.

Eqs. (14) and (15) represent the PNB as it is related to surplus, harvested, and total applied nitrogen:

$$\mathrm{PNB}_{gi} = 1 - \frac{\mathrm{SurN}_{gi}}{\mathrm{AppN}_{gi}} = 1 - \frac{\mathrm{AppN}_{gi} - \mathrm{HarN}_{gi}}{\mathrm{AppN}_{gi}} = \frac{\mathrm{HarN}_{gi}}{\mathrm{AppN}_{gi}} \tag{14}$$

$$\mathrm{HarN}_{gi} = \mathrm{PNB}_{gi}\mathrm{AppN}_{gi} \tag{15}$$

where SurN_{gi} = nitrogen surplus; and HarN_{gi} = nitrogen removed by harvest. It is also assumed that irrigation efficiency exceeds or equals the PNB, as some farming operations may, for example, have well-managed drip irrigation with a high water use efficiency, but still have a low PNB from remaining inefficient nitrogen management.

It is assumed that a high PNB cannot occur when irrigation efficiency is low. However, there may be events or seasonal cases when irrigation efficiency is poor, yet nitrogen leaching is also low. This may occur, for example, if soil nitrate concentration is low during preirrigation or during the winter (rainy season), when groundwater recharge is high. Likewise, reducing soil nitrogen during these times is an improved practice. This approach considers annual groundwater N loading, with particularly concern for irrigation season losses.

Policy Simulations for Nitrogen Use Efficiency

Policy simulations estimate changes in agricultural revenues from shifts in crop patterns (including increased fallowing) attributable to nitrogen load reduction policies. Also estimated are changes in revenue from efficiency improving management, taxes on nitrogen use, maximum load limits, and other policies. The authors are not concerned here with specific aspects of such policies or with the political feasibility of these policies. Instead, the focus is on expected shifts in cropping patterns and changes in farm revenues at different levels of restrictions on nitrogen leaching.

For some of the policy scenarios, restrictions are added on the total nitrate load to groundwater in a region g. For this, a new constraint is added that limits the total nitrogen load to a fraction, Red_g, of the base total groundwater nitrate load in region g:

$$\sum_i \mathrm{GW}_{\mathrm{load},gi} \le \mathrm{Red}_g \sum_i \tilde{X}_{gi,\mathrm{AppN}}(0.9 - \mathrm{PNB}_{gi}) \qquad (16)$$

On the left side, groundwater load for region g is as in Eq. (10); Red_g is the policy determined factor to reduce loading to groundwater by some percentage for region g, and the summation over i is the *current* groundwater nitrogen load from crop i in region g, assuming that from the *observed* applied nitrogen ($X_{gi,\mathrm{AppN}}$tilde) 10% is lost to atmosphere and the rest is removed by harvest. Thus Red_g would equal one unit for a base case with no reductions, and 0.75 if nitrogen load to groundwater is reduced by 25%. Water use efficiency is constrained to exceed PNB, such that the weighted PNB is less than the weighted regional water use efficiency.

In summary, the process has five steps (Medellin-Azuara et al. 2012a): (1) linear land constrained program [Eqs. (1)–(5)]; (2) estimation of a calibration PMP cost function; (3) parameterization of the irrigation efficiency nest; (4) parameterization of the nitrogen efficiency nest; and (5) base calibrated model [Eqs. (6)–(11)]. In this fifth step, regional producers' surplus [Eq. (6)], tradeoffs between costs, and efficiency in irrigation and nitrogen management [Eqs. (7) and (8)] are maximized, with constraints on resources [Eq. (9)], mass balance [Eq. (10)], and policy-based nitrogen leaching limits [Eq. (13)].

Model Data Sets

The model is calibrated on the basis of publically available data sets. Because data are insufficient to estimate a baseline and improved irrigation and nitrogen set of practices for all crops in the two study regions, the analysis is performed on crop groups, aggregating crop groups data on the basis of area-weighted averages. One shortcoming of this crop group approach is aggregation of the response: all crops in a group are assumed to respond equally to costs of improvement.

Irrigation and Cost Data

Production input use of land, water, labor, and supplies (excluding nitrogen) are from the Statewide Agricultural Production Model (SWAP) (http://swap.ucdavis.edu, Howitt et al. 2012). Irrigation efficiency, the ratio of evapotranspiration of applied water to applied water, was taken from the California 2009 Water Plan (http://www.waterplan.water.ca.gov/). The capital costs per unit area for irrigation efficiency were from Hatchet (1997) and scaled to 2008 dollars, as were other monetary costs on inputs. Production information from University of California Davis agricultural cost and return studies was used for additional crops, including lettuce and strawberries (http://coststudies.ucdavis.edu/). The irrigation technology parameters for the CES trade-off curves follow Hatchett (1997).

Nitrogen Use and Cost Data

Because data are generally unavailable to estimate nitrogen use or cost data for individual practices, yet alone bundles of practices, data sets were developed to estimate efficiency and costs for three scenarios of practices: a current baseline scenario (Bundle 1), an improved scenario (Bundle 2), and an idealized and most efficient scenario (Bundle 3). Bundle 1 represents the efficiency and cost of current practices. Bundle 2 represents the scientifically tested improvement in nitrogen management possible with currently available practices. Bundle 3 represents the presumed benefits for PNB, surplus, and nitrogen loading and economic costs for practices that are under development or not yet practically feasible at scale.

PNB of Bundles

The first step in developing the data set was to estimate the PNB for each of the three bundles. For Bundle 1, representing baseline or current practice, PNB was calculated from available statistics. Calculating a PNB required yield (USDA 2011b), moisture and nitrogen content of the crop (USDA 2011a), and nitrogen application rates (Rosenstock et al.). Because the PNB of Bundle 1 reflects statewide average reported values, it aggregates across all current practices. This includes both advanced nitrogen management practices in some cases, as well as more traditional nitrogen management practices in others. The PNB derived from the statewide averages is set to equal the PNB for the most common unimproved bundles. Implicitly this means that depending on the current extent of adoption of improved practice bundles, the baseline PNB may be underestimated.

Bundle 2 includes the so-called "improved," scientifically verified, practices. The PNB data for this bundle were compiled through a review of published literature and collected unpublished data on the most recent research on nitrogen management in California for 22 economically important crops (Dzurella et al. 2012). These studies and data reflect recently developed and tested nitrogen and irrigation best management practices. The authors sought to include research from field-scale nitrogen trials. Research station results were excluded when other research existed because PNB tends to be higher under research-station conditions than in grower's field. Where research reported the nitrogen in the harvested portion of crop, those values were used directly. Where research only reported yield, but not crop nitrogen content, the amount of nitrogen in the crop was calculated on the basis of the USDA Crop Nutrient Tool (USDA 2011a).

Bundle 3 represents the highest plausible gains. Many practices currently used by growers or under development, such as weather-based irrigation scheduling in cool-season vegetables, will potentially reduce nitrogen loading further than the improved practices previously identified. However, data quantifying PNB and nitrate loading are not available. These most efficient practices are represented by including a third hypothetical bundle in which it is assumed that PNB is 5% higher than in the improved practice bundle.

Costs of Bundles

Estimated costs of bundles are unavailable, especially for the range of crops grown in the study regions. Because of the paucity of data, an index to estimate costs and the differences in costs among the bundles (referred to as the cost ratio) was developed. The cost ratio estimates the relationship between the cost of applying fertilizing materials—e.g., labor, machine time, information—and the fertilizing materials themselves. The cost ratio is based on the assumption that improving PNB generally results from more active management, demanding more resources. As nitrogen and water management improve, application cost increases relative to purchase cost.

Cost ratios for the baseline and improved scenarios for each crop group were derived from the University of California Agricultural and Resource Economics Department Cost and Return Studies (CS) (http://are.ucdavis.edu). Estimated costs of bundles were developed to be consistent with the agronomic practices used to calculate PNB (e.g., industry standard practices for Bundle 1 and the practices used in nitrogen trials for Bundle 2). Details on the cost standardization from cost and return studies are presented in Dzurella et al. (2012). Because the created ratios of costs were consistent within a study, the ratios are comparable across studies. These cost ratios are employed to estimate the CES relationship between nitrogen use efficiency and investments in nitrogen use efficiency. Improvements in PNB modeled in this study lay within the continuum of this entropy-estimated relationship, shown graphically in Dzurella et al. (2012).

Modeling Results and Discussion

Policy Modeling

The California version of the Federal Clean Water Act of 1972 (CWA), the so-called Porter-Cologne Act of 1969 (PCA), is the governing regulatory framework controlling discharges of nitrate to groundwater. Although groundwater contamination is not regulated by CWA, California's PCA specifically includes groundwater and is overseen by the State Water Resources Control Board. Their Regional Water Boards implement both federal and state authority through NPDES permits, regional plans, the dairy waste discharge requirements (WDR) regulatory program for the Central Valley, the Irrigated Lands Regulatory Program, and other programs. Some of these programs have established or are expected to establish guidelines for reporting and monitoring nitrogen use by agricultural producers. In some cases, water discharge requirements (a type of permit) are issued. The modeling approach discussed in this paper allows for a quantitative analysis of the costs to agricultural producers from regulatory requirements on total nitrogen (nitrate) loading to groundwater, such as taxes, penalties, cap-and-trade, outright restrictions on applied nitrogen, or performance standards on groundwater loading with nitrogen. Such analysis allows for promising policy options to reduce nitrate contamination. Results can provide a reference in designing regional water quality programs or identifying revenue generating schemes to mitigate nitrate contamination problems.

Two baseline crop mixes were modeled—one for the Tulare Lake Basin and one for the Salinas Valley—considering the so-called Nitrogen Hazard Index grouping (Harter et al. 2012). The hazard index indicates nitrate leaching vulnerability on the basis of soil characteristics, the crop grown, and the irrigation system for a specific field. Similar approaches have been used to quantify vulnerability of groundwater in agricultural regions (Loague et al. 1996). This work employed existing cost information in the SWAP model and information on the likely PNB and its cost before and after application of best management practices. The model calibrated for all selected crops and production factors to within 3% of the observed input values.

Several policy scenarios were modeled:

- Cap-and-trade scheme limiting the total nitrogen load to groundwater within a region. Two different caps were implemented: a 25% reduction and a 50% reduction in total nitrogen load to groundwater within a region (no limit on the local nitrogen load to groundwater). This is similar to performance standards modeled in previous work by Johnson et al. (1991).
- Tax on applied nitrogen. Two different tax scenarios were implemented: a tax of 7.5%, which is equivalent to a sales tax; and, via iterative optimization, the tax level determined to be necessary to reduce the total nitrogen load to groundwater within a region by 25% (no limit on the local nitrogen loading to groundwater).
- A surcharge per kilogram of nitrogen of \$4.4 for crops exceeding a groundwater load of 35 kgN/ha. This policy partially mimics the Netherlands Mineral Accounting System that has shown some success in reducing nitrogen surplus in agricultural areas (Canada et al. 2012). However, here the surcharge is fixed on the basis of the current, not future, groundwater nitrogen load (no limit on the local nitrogen load to groundwater).
- A cap and trade on applied nitrogen, in which the cap is optimized through iterative optimization such that the total nitrate load to groundwater in a region is reduced by 25% (no limit on the local load to groundwater).
- A prescriptive performance standard on applied nitrogen such that all land in agricultural production does not exceed threshold levels for nitrate loading to groundwater (the maximum local load to groundwater is fixed everywhere). Two thresholds were chosen: 35 and 70 kgN/ha/year. The first threshold is commensurate with proscribing that groundwater recharge contains no more than about 45 mg/L nitrate (drinking water standard), given typical regional groundwater recharge rates. The second threshold allows for twice that level.

In all policy scenarios, yields are assumed to be constant. For each scenario, changes in land use (production levels), and changes in revenues (costs) are computed. The robustness of the approach is tested by a sensitivity analysis of the marginal cost of improving nitrogen use efficiency.

In previous work, tax and other fee-based N reduction policies have the highest social costs (Helfand and House 1995); however, these uniform input taxes and regulations (same for all users) are close to the socially optimal solution when accurate pollution charges are difficult to implement. The effect of uniform regulations for agricultural production was quantified in the study area.

Modeling Results

All modeled policy scenarios assume that adjustments occur in land use and management practices, whereas yields are maintained by improving nitrogen and irrigation efficiency. The model results in-

Table 1. Cap-and-Trade Groundwater Nitrogen Load Reduction Scenarios and Associated Changes in Total Applied Water, Annual Net Revenues, Irrigated Land Area, and Applied Nitrogen

Region	Scenario	Applied water, hm^3/year	Net Revenues $2,008 M/year (%)	Irrigated land, 1,000 ha	Applied nitrogen 1,000 ton/year (%)
Tulare	Base load	10,530	4,415	1,293	200
	25% reduction	10,134 (−3.7%)	4,259 (−3.5%)	1,240 (−4.1%)	181 (−9%)
	50% reduction	7,830 (−29%)	3,783 (−14%)	952 (−26.4%)	135 (−32%)
Salinas	Base load	366	309	92	18
	25% reduction	328 (−10.4%)	285 (−7.5%)	83 (−9.7%)	15 (−16%)
	50% reduction	246 (−32.8%)	239 (−22%)	62 (−32.6%)	10 (−46%)

Note: The model is constrained to keep yield for each crop constant.

dicate a relatively mild economic adjustment in cultivated land for 25% nitrate load reduction. However, a 50% reduction from the current nitrate load across the entire region translates into higher production costs and some decreases in net revenues from farming (see Table 1). A target average reduction over the region implies a regional market for nitrogen leachate among farmers and fields similar to a cap and trade scheme (Canada et al. 2012). Base water use efficiencies are roughly 70% for all crops in Tulare Lake Basin and Salinas Valley. Baseline nitrogen use efficiency for the crops analyzed is 51% for TLB and 40% for SV. Tulare Lake Basin has a higher proportion of more nitrogen-efficient crops such as corn, processing tomatoes, almonds, and pistachios; however, the model does not account for manure as a soil amendment.

Both water and nitrogen use efficiency increase with restrictions on total nitrogen load to groundwater. The marginal cost of increasing irrigation efficiency exceeds the marginal cost for increasing nitrogen efficiency. Thus the model allocates fewer resources to improve irrigation efficiency. To represent the interaction of water and nitrogen use efficiency on nitrate percolation to groundwater, the model was constrained such that water efficiency always equals or exceeds nitrogen application efficiency.

The initial reductions of 25% of the deep percolation load result in relatively small reductions in net farm revenue (see Table 1). This assumes some education in farm management regarding nitrogen management practices. However, net revenue losses increase at an increasing rate as greater reductions are sought. On average in the TLB, a reduction of 3.6 t of applied nitrogen for every 405 ha (1000 acres, or 8.1 kg/ha) must be in place to achieve a 25% decrease in regional load to groundwater. For the Salinas Valley, this reduction is close to 5.7 t per 405 ha (12.8 kg/ha). For a 50% reduction in load to groundwater, the required reduction per 405 ha increases to 5.2 t for the TLB and 12.9 t for the Salinas Valley (respectively, 11.6 and 28.9 kg/ha). Nitrate load reductions are also achieved by land fallowing (see Table 1). In the TLB, losses in net revenue at 50% reductions in nitrogen load to groundwater are estimated as 14%, four times the loss of a 25% reduction (3.5%). A similar relationship holds for Salinas (see Fig. 2).

PNB increases with policies that restrict nitrogen loading to groundwater. At base load conditions, average weighted PNBs of 0.51 and 0.40 are estimated for TLB and SV, respectively. If a 25% reduction in the N load to groundwater is implemented, weighted average PNB increases to 0.58 and 0.44 for TLB and SV, respectively. The ratio of applied nitrogen to effective nitrogen decreases under nitrogen load to groundwater restricting policies. Conversely, the ratio of investments in PNB to effective nitrogen increase as nitrogen load to groundwater is restricted. Changes in these ratios suggest farming adaptation to N groundwater load reduction policies by reducing applied nitrogen, increasing PNB via investments in technology bundles, reducing irrigated crop areas, or switching to more nitrogen efficient and profitable crops.

Net percentage revenue reductions are shown in Fig. 2, corresponding to net revenues in Table 1. Net revenue losses increase rapidly with larger reductions in total nitrogen load to groundwater. Reductions in average nitrate load to groundwater of 25% have an

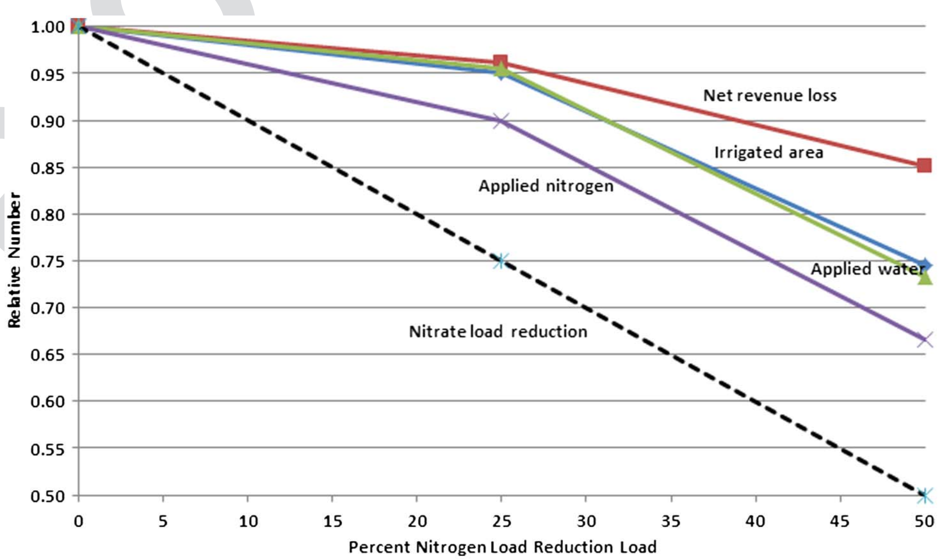

Fig. 2. Relative applied water, net revenues, irrigated land, nitrogen applied, and nitrogen load reduction as a function of reductions in the nitrogen loading to groundwater by agriculture averaged over the Tulare Lake Basin and Salinas Valley

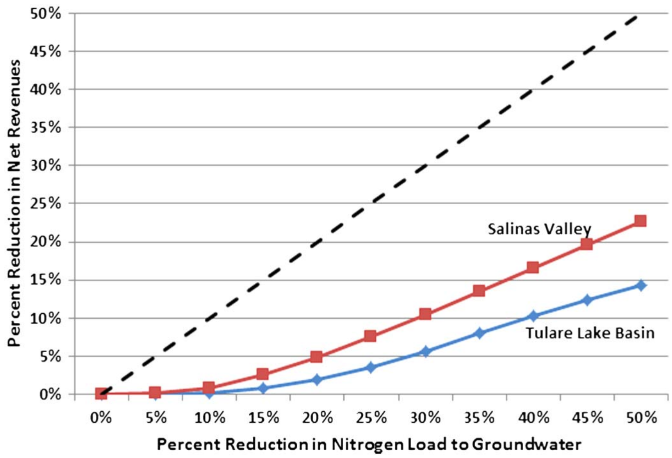

Fig. 3. Percentage reduction in net revenues estimated from different levels of reduction in nitrogen loading to groundwater

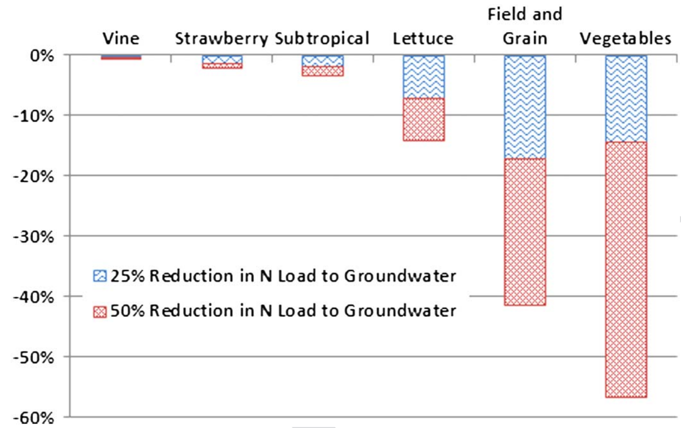

Fig. 4. Cumulative change in cropping patterns with respect to base conditions for selected crops in Salinas Valley

average revenue reduction of \$8.1 per kilogram of applied nitrogen in the surface in TLB and SV. When average nitrate load to groundwater is reduced by 50%, the average cost of reduced applied nitrogen is approximately \$9.7 per kilogram for TLB and \$9.1 for SV. Both revenue reductions account for less irrigated land area.

The average net revenue loss per kilogram of nitrogen load to groundwater is roughly \$8/kg when the total nitrate load to groundwater is reduced by 25%. At the 50% reduction level, the marginal net revenue loss per kilogram of nitrate load reduction is \$18/kg, nearly twice as much as the average net revenue losses at 25% N load reduction. This includes the revenue losses attributable to land fallowing. The cost per unit of applied nitrogen reduced increases less between 25 and 50% nitrate load reductions than the net revenue losses. The reason is that, over this range, adjustments also occur in the amount of applied nitrogen, which results in applied nitrogen being reduced more than the net revenue losses (see Table 1) because of changes in its application intensity, cropping patterns, and land fallowing.

The resulting cropping pattern changes from the two nitrate loading reduction levels by crop group was estimated. With higher reductions (50%), cotton, corn, and other field and grain crops have the largest reductions in the Tulare Lake Basin. Irrigated field and grain crop area is also reduced in the SV (see Fig. 4) where higher value crops are grown instead. Irrigated area for high-value crops such as strawberries and lettuce remain about the same. However, vegetable crops as a group, because of their lower PNB, have reduced crop area at higher restrictions of nitrate load to groundwater.

Tax on Applied Nitrogen

A tax on nitrogen use is one way to simultaneously reduce nitrogen use and raise revenues for alternative water supplies. Currently, commercial nitrogen fertilizer sales in California are not subject to sales tax. The economic model is run for the case where purchase of nitrogen is subject to the standard 7.5% sales tax. Under this tax, the model predicts that farmers will respond in several ways to minimize the costs of the tax. There is a small difference in revenues and reductions in the levels of nitrogen applied in response to cost increase. Savings in fertilizer expenses are mostly offset by increases in investment in improving nitrogen use efficiency. The tax reduces overall nitrogen application by roughly 1.6% for both basins, an elasticity close to that found by Johnson et al. (1991). Total irrigated acreage remains almost unchanged. For tax rates on applied nitrogen below 50%, the relationship between net revenue losses and tax rate is nearly linear. Cropping patterns are similar to base conditions. Net revenue losses for both TLB and the SV from a sales tax policy of 7.5% are close to \$29.4 million (0.6% of base net revenues) and tax revenues are \$27 million, for a net welfare loss of \$2.4 million per year.

Penalty for Nitrogen Use above a Threshold

A penalty of \$4.4 per kilogram of nitrogen use for crops exceeding an average load of 35 kg/ha (32 lb/acre) is examined. Under this policy, irrigated area is reduced by 4.5% in the TLB and 5.6% in the SV. Total revenue losses were 2.3 and 4.4% for TLB and SV, respectively, slightly less than the irrigated crop area reductions. However, net revenue losses in this case exceed percentage land use reductions. Nearly 20 and 26% reductions in net revenues can be expected for TLB and SV, respectively. This is attributable to much higher fertilizer costs in high-value crops with low PNB, e.g., vegetables, and crop shifts.

Comparing Policies to Achieve 25% and 50% Nitrogen Load Reductions to Groundwater

As in previous studies (Johnson et al.1991), the effects of a 25% nitrogen load reduction over a wider range of policy options were examined. A constant nitrate load reduction (25%) was maintained, and changes in net revenues, applied nitrogen, irrigated area, and applied water were estimated. Policy options include cap and trade on nitrate load, a cap-and-trade scheme on applied nitrogen, a tax on applied nitrogen, a mandated efficiency improvements program (technology standard), and a loading limit as a performance standard. The performance standard that limits the nitrate load to groundwater to no more than 70 kg N/ha also yields overall groundwater load reductions of about 25%. Table 2 shows a summary of the impacts from these policies. Also included for comparison is the 50% cap-and-trade policy on groundwater load reduction and the 35 kg N/ha performance standard policy, which yields a similar overall nitrogen load reduction to groundwater of about 55%.

Results show an overall similar level of reduction in applied nitrogen, irrigated area, applied water, and net revenues for comparable levels of regional groundwater load reductions, regardless of policy. A tax on nitrogen of nearly 150% for TLB and 185% in SV is required to achieve the 25% load reduction. The tax policy shows the highest net revenue loss. The highest reductions in applied

Table 2. Summary of Impacts from Various Policy Scenarios that Achieve an Overall 25% (5 Scenarios) or 50% Nitrate Load Reduction in Crop Agriculture (2 Scenarios)

Scenario	Cap and trade NO_3 load to groundwater (see Table 1)	Cap and trade on applied N	Tax on applied N	N and water efficiency investment	Prescriptive[a] performance standard on NO_3: load: 70 kg/ha	Cap and trade NO_3 load to groundwater (see Table 1)	Prescriptive[b] performance standard on NO_3: load: 35 kg/ha
Action level (%)	−25	−9.3 TLB	147 TLB	9.6 TLB	Variable[a]	−50	Variable[b]
		−12.5 SV	185 SV	13.9 SV			
Applied N change (%)	−9 TLB	−9.2 TLB	−10.3 TLB	−9.2 TLB	−9.2 TLB	−32 TLB	−26.2 TLB
	−16 SV	−12.6 SV	−16.9 SV	−16 SV	19.3 SV	−46 SV	−36.1 SV
Irrigated area change (%)	−4.1 TLB	−2.9 TLB	−3.7 TLB	−3.1 TLB	−3.1 TLB	−26.4 TLB	−20.2 TLB
	−10.7 SV	−8.1 SV	−9.6 SV	−8.3 SV	−8.4 TLB	−32.6 SV	−28.7 SV
Applied water change (%)	−3.9 TLB	−4.6 TLB	−3.2 TLB	−6.4 TLB	−5.3 TLB	−29 TLB	−21.6 TLB
	−11.6 SV	−13.5 SV	−9.1 SV	−11.1 SV	−15.8 SV	−32.9 SV	−29.7 SV
Net revenues change (%)	−3.5 TLB	−3.3 TLB	−4.9 TLB	−3.8 TLB	−3.9 TLB	−14 TLB	−12 TLB
	−7.5 SV	−7.2 SV	−11.7 SV	−7.9 SV	−9.4 SV	−22 SV	−20 SV

Note: TLB = Tulare Lake Basin; SV = Salinas Valley.
[a]This is a performance standard that yields 25% NO_3 load reduction for the TLB and 31% for the SV; in this scenario, load reductions for individual crops vary, but all crops achieve the performance standard.
[b]This is a performance standard that yields 55% NO_3 load reduction for the TLB and for the SV.

water and nitrogen are obtained with investments in nitrogen and water use efficiency. With higher PNB, less land is put out of production to meet the 25% load reduction across each region. A cap and trade on applied nitrogen shows similar performance to a cap and trade on nitrate load, which is not surprising, as the two are directly related through the PNB. A cap and trade on applied nitrogen would be preferable over a cap and trade on groundwater nitrate loading, which would be much more difficult and expensive to monitor for compliance than monitoring applied nitrogen. Results indicate that rules on applied nitrogen can successfully be applied to control groundwater nitrate loading.

Furthermore, at this large scale, the model is not directly accounting for heterogeneity in soil conditions, which may also drive leaching under either policy. Nevertheless, the difference in the net revenue changes with each policy (columns 2 and 3) is marginal. The prescriptive performance standard is the only policy investigated that guarantees basin-wide compliance with groundwater loading limits. Despite its prescriptive nature, the overall changes in water and nitrogen management and the associate costs (revenue losses) are nearly identical to those under the cap-and-trade programs or the technology standard approaches. Achieving an average load of 35 kg/ha, thus guaranteeing drinking water quality, for all land in agricultural production across both TLB and SV yields similar changes in management practices. The amount of land fallowing and the net revenue changes are again comparable to a cap-and-trade system of similar regional groundwater load reduction, while guaranteeing more uniform compliance with groundwater quality standards. In either case, large revenue losses are expected to occur.

Sensitivity Test

One of the most uncertain parameters is the marginal cost of improving nitrogen use efficiency. The sensitivity of model results to this cost assumption is tested by doubling this cost in the model. The model is then calibrated using the higher marginal cost and the same elasticities of substitution and supply as the base results model. When coupled with this higher marginal cost for improving PNB, the 7.5% tax reduces both nitrogen applied and irrigated land area by 2.3% in the TLB and 3.2% in the SV. The higher cost of using a technology to improve nitrogen use efficiency makes it less expensive to reduce irrigated crop acreage of some crops than to adopt the efficiency enhancing practices. Crop area reductions occur for field crops and corn both in the TLB and in the SV are less than 10% of the base cultivated land. Net revenues decrease by 10.4% in the TLB and 15.3% in the SV. For the nitrogen load to groundwater restriction policies, at a 25% reduction, irrigated crop area decreases 10% for TLB and 15% for SV. This sensitivity analysis confirms that the cost at which substitution between capital required to improve PNB, and the resulting PNB, is a critical parameter for both the modeled cost and type of policy response.

The range over which best practices for applied nitrogen can be substituted is critical to the costs of both policy scenarios. The authors had great difficulty in finding reliable measures of the ability to substitute application technology for applied nitrogen in the agronomic literature. Additional research on this topic is required to more reliably model the cost of nitrogen reduction policies.

Limitations

Several limitations are worth noting in this modeling approach. First, the aggregation of crops may bias crop farming response to nitrogen load limiting policies in both directions. Load limits and reductions are averaged over large areas, so local reductions in nitrate loads could vary greatly with local cropping and other decisions. Second, the restriction that keeps yields constant will overestimate the cost of both nitrogen load limiting and nitrogen cost policies as higher PNB may increase yields and therefore increase gross farming revenues (Hartz 1994). Third, carryover nitrogen (Knapp and Schwabe 2008) and crop rotation may influence multiyear cropping decisions currently not modeled, which may overestimate the cost of policies modeled here. Fourth, given California's market power for some specialty crops, irrigated crop area shifts may have some endogenous price effects that influence production decisions that might also reduce the estimated revenue losses. A more comprehensive approach to capture price effects could be used (Medellin-Azuara et al. 2012b). Finally, the interaction of applied water and nitrate on the load to groundwater is often explicitly modeled (Johnson et al. 1991; Wu et al. ~~1994;~~ Knapp and Schwabe 2008). In the present long-term mass balance approach, this interrelationship is based on the efficiency rates for both irrigation and nitrogen use. With these limitations in mind, this approach is useful in estimating likely crop response and costs of nitrogen use efficiency management for California.

Conclusions

Consistent with the literature (Knapp and Schwabe 2008; Larson et al. 1996; Vickner et al. 1998), small reductions in nitrogen leaching to groundwater can be made at relatively low costs. Adjustments occur in three ways, including changes in nitrogen use efficiency, changes in irrigation efficiency, and changes in cropping patterns (including reduction of irrigated area). The response to policy measures is sensitive to both the cost of increasing nitrogen use efficiency and the range over which improved efficiency can substitute for applied nitrogen. In constructing the model, the ability and cost of improving nitrogen use efficiency is difficult to define quantitatively, given current agronomic studies and available data. The marginal cost of increasing nitrogen use efficiency is the most critical parameter in terms of uncertainty, and should be the focus of additional empirical field studies such as those done for irrigation efficiency, before policies are based on results such as these.

Several conclusions arise from this work:

1. Modest increases in nitrogen use efficiency will increase production costs but are unlikely to affect total irrigated crop area. Less than 4% of the total irrigated area and net revenues will be lost with modest increases to PNB through improved management practices.
2. Larger reductions in excess nitrogen will be much more costly and may lead to reductions in irrigated area, lower net revenue, and shifting cropping patterns towards more nitrogen-efficient crops. For large reductions in excess nitrogen, more than 20% of total irrigated grain and field crops area would be reduced in both basins.
3. A sales tax on applied nitrogen may slightly decrease total applied nitrogen with some loss in farm net revenues. A sales tax of 7.5% could help reduce applied nitrogen by nearly 2% under the modeling and cost assumptions developed here.
4. If the marginal costs for increasing nitrogen use efficiency are larger than estimated, farm costs for nitrogen limiting and tax policies will increase. Doubling the marginal cost of improving nitrogen use efficiency reduces net revenues more than 14% in the TLB and 21% in the SV when total nitrogen loading is reduced by 25% of base values.
5. A prescriptive standard on nitrogen load across the entire two groundwater basins yielded similar fallowing acreage and farm revenue losses to a free market cap-and-trade approach with similar total groundwater load reductions. The prescriptive standard will cap nitrogen load to groundwater across the entire region, whereas a cap-and-trade approach allows for local groundwater pollution with large N loads, balanced by much cleaner recharge elsewhere.

Combining quantitative economic and agronomic data into a regional level model can provide insights into the costs and other consequences of policy alternatives designed to achieve reductions in ground water nitrogen load.

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Notation

The following symbols are used in this paper:

a_{gij} = Leontief coefficient of production input j for crop i in region g;
b_{gj} = Available amount of resource j in region g;
g = Region, Tulare Lake Basin (TLB) and Salinas Valley (SV);
i = crop group, following Dzurella et al. (2012): alfalfa, almonds and pistachios, corn, cotton, grain and field, lettuce, orchards, strawberries, subtropical, tomato, vegetables, and vine crops;
j = Production input: land, effective water, effective nitrogen, investments in water use efficiency, investments in nitrogen use efficiency, applied water, applied nitrogen and supplies;
V_{gi} = Price per yield crop yield in (dollars per t) for crop i in region g;
$\tilde{X}_{gi}$ = Observed (base) amount of input j in region g for crop i;
XL_{gi} = Decision variable for land in the first stage production function program for crop i in region g;
XN_{gij} = Decision variable for input j in the nested CES production function program for crop i in region g;
XNN_{gij} = Decision variable for input j in the main CES production function of the final stage for crop i in region g;
Y_{gi} = Yields per unit area (t/Ha) for crop i in region g;
$\beta 1_{gi}$ = Share parameters for the nested water use efficiency CES function for crop i in region g;
$\beta 2_{gi}$ = Share parameters for the nested nitrogen use efficiency CES function for crop i in region g;
β = Share parameters for the main CES production function for crop i in region g;
γ_{gi} = Slope parameter in the marginal PMP quadratic cost function;
δ_{gi} = Intercept parameter in the marginal PMP quadratic cost function;
$\lambda 1_{gij}$ = Lagrangian multiplier of the calibration constraint for region g, crop i and input j;
$\lambda 2_{gij}$ = Lagrangian multiplier of the resources constraint for region g, crop i and input j;
τ_{gi} = Scale parameter of the main CES production function for region g and crop i;
τ_{Ngi} = Scale parameter of the nested nitrogen use efficiency CES function for region g and crop i;
τ_{Wgi} = Scale parameter of the nested water use efficiency CES function for region g and crop i; and
ω_{gij} = Linear cost of production input j for crop i in region g.

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Queries

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