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Controlling Dairy Nitrogen Emissions:
A Dynamic Analysis of Herd Adjustment, Ground Water Discharges, and Air Emissions

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Kenneth A. Baerenklau
Assistant Professor
Department of Environmental Sciences
420 Geology Building
University of California
Riverside, CA 92521
Phone: (951) 827-2628
Email: ken.baerenklau@ucr.edu

Nermin Nergis
Postdoctoral Researcher
Department of Environmental Sciences
415A Geology Building
University of California
Riverside, CA 92521
Phone: (951) 827-2875
Email: nermin.nergis@ucr.edu

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Abstract

A micro-dynamic model of a livestock-crop operation is calibrated with data from a representative dairy in California's Central Valley and is used to predict the effects of regulations designed to reduce nitrogen emissions. Policy simulations clarify the importance of dynamic elements and demonstrate three main results: (1) dairies are unresponsive to pollution charges unless they are relatively large and financially burdensome for farmers; (2) regulations aimed at controlling only nitrate leaching will cause significant increases in ammonia emissions; and (3) mitigating both nitrogen problems with emissions taxes involves substantial reductions in both herd size and farm profit.

Key Words

Cross-media pollution, dairy, dynamic optimization, emissions taxes, groundwater, nitrogen.

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Nitrate contamination of ground water continues to be a problem throughout the United States. Nationally, approximately 22% of domestic wells in agricultural regions exceed the federal maximum contaminant level for nitrate (Ward et al. 2005). In California, between 10 and 15% of water supply wells exceed the standard (Bianchi and Harter 2002). To better understand this problem and evaluate policy outcomes, recent economic studies of nitrate leaching from crop production have focused on micro-level decisions (Ekman 2005 and citations therein) and have given explicit treatment to dynamic and/or spatial aspects of agricultural production (Schwabe and Knapp 2005; Anselin, Bongiovanni and Lowenberg-DeBoer 2004; Khanna, Isik and Winter-Nelson 2000; Vickner et al. 1998; Watkins, Lu and Huang 1998). In a recent example that incorporates all of these elements, Schwabe and Knapp (2005) develop a multi-period farm-level model of corn production that accounts for forward-looking behavior, soil nitrogen dynamics and spatially non-uniform irrigation. Their results demonstrate the importance of the spatial component: leaching rates with typical irrigation system non-uniformity are five to six times larger than with uniform irrigation. But the importance of the dynamic components is not as great, largely due to the inherently fast transition to steady-state levels of soil nitrogen.[1]

This literature is relevant for livestock-crop operations, as well. Livestock-crop producers, such as dairy farmers, convert feed into market goods (milk and meat) and byproducts (manure). The manure is used to fertilize crops (or sometimes exported from the farm) which are then used as feed, and the cycle continues. Nitrogen problems – not only nitrate leaching but also ammonia volatilization – arise from the same circumstances faced by crop-only producers (i.e., free disposal) and are exacerbated by the economics of livestock production. In particular, dairies tend to exhibit non-decreasing returns to scale at traditional operating levels (Moschini 1990). This has led to the proliferation of modern dairies with thousands of animals – compared to traditional stocking rates of a few hundred animals – in places like the California's Central Valley. These concentrated animal feeding operations (CAFOs) generate significantly more manure nitrogen than can be assimilated by locally grown crops (Gollehon et al. 2001). Because manure is costly to transport (Ribaudo et al. 2003), because other farmers' willingness to accept manure as a substitute for commercial fertilizer is not very high (Norwood, Luter and Massey 2005), because effective waste treatment is expensive (SJVDMTFAP 2005) and because regulations have been largely ineffective (Dodd et al. 2003; BVA 2003) dairy farmers dispose of nitrogen by over-applying manure to crops and by allowing large amounts of ammonia to volatilize. This has produced nitrate leaching rates from dairy fields that are ten times greater than at crop-only operations (VanderSchans 2001; Pang, Letey and Wu 1997) and has contributed to high concentrations of fine airborne particulate matter in dairy-intensive airsheds.[2]

In response to these problems and to the changing nature of the dairy industry, the United States Environmental Protection Agency (USEPA) recently issued revised guidelines for CAFO emissions to surface and ground water (USEPA 2003). State-level agencies that oversee dairy-intensive regions, such as the Central Valley Regional Water Quality Control Board and the San Joaquin Valley Air Pollution Control District, also are pursuing more effective regulations though typically not in coordination with one another.

Research by agricultural economists continues, as well. Although several authors have addressed livestock waste problems with regional or static models (Kaplan, Johansson and Peters 2004; Ribaudo, Cattaneo and Agapoff 2004; Feinerman, Bosch and Pease 2004; Innes 2000; Alireza, Howitt and Mariño 1994), the existing literature contains few examples of micro-dynamic models of livestock-crop operations that can be used to predict farm-level responses to environmental regulations through time (an exception is Schnitkey and Miranda (1993), who examine phosphorus regulations and spatial patterns of manure application). The importance of dynamic aspects should not be dismissed for the case of livestock-crop production. Unlike crop-only production, as examined by Schwabe and Knapp (2005), a livestock-crop producer invests in a capital stock of animals that evolves through time, becomes obsolete, generates a salvage value and is replaced. Managing this capital stock introduces additional dynamic elements that are not present in the crop-only problem and that may tend to delay producer responses to changing operating conditions. Previous studies have incorporated such dynamic elements when examining farm management decisions (e.g., Tozer and Huffaker 1999; Chavas and Klemme 1986; Van Arendonk 1985) but not in the context of environmental pollution control.

In this article we develop a micro-dynamic model of a concentrated animal feeding operation with crop production and non-uniform irrigation, calibrate it with farm-level data from a well-documented dairy in California's Central Valley, and use it to predict the effects of regulations intended to control nitrogen emissions from modern dairy farms. Our policy simulations clarify the importance of incorporating dynamic elements into models of livestock-crop operations and demonstrate three main results: (1) dairies are unresponsive to pollution charges unless they are relatively large and thus financially burdensome for farmers; (2) regulations aimed at controlling only nitrate leaching will cause significant increases in ammonia emissions; and (3) mitigating both nitrogen problems with emission taxes involves substantial reductions in both herd size and farm profit. When viewed in light of political concerns, these results suggest that raising the cost of nitrogen disposal is not, by itself, a feasible solution to the multi-media dairy nitrogen problem. Subsidizing on-farm waste treatment technologies and/or mitigating ambient pollution levels at receptor points may be necessary, as well.

A Dynamic Model of a Dairy Farm

Herd Management

Our model farmer works in discrete time, measured in years, and manages a self-replacing herd of Holstein calves, heifers and milk cows. For simplicity we assume animal characteristics differ across age cohorts but not within cohorts.[3] Each year the farmer decides how many animals from each cohort to retain and how many to sell (cull). Assuming a ten-month calving cycle followed by a two-month dry period (Chang et al. 2005), a normal healthy cow would spend her first year on the farm as a calf and her second year as a bred heifer before calving for the first time at the end of her second year. She would then calve at the end of each subsequent year until she is culled. Assuming each cow is culled no later than the end of her fifth year (Tozer and Huffaker 2000) gives a herd age distribution with five discrete intervals. Incorporating a calving rate, a mortality rate, the option to purchase replacement heifers, and assuming all cows that do not calve are culled, the equations of motions for the herd age cohorts are given by:

$$H_{a,t+1} = \begin{cases} \sum_{\tilde{a} \in \{2,3,4\}} \theta_{\tilde{a}t} \gamma^f \gamma^b_{\tilde{a}} \gamma^s_{\tilde{a}} H_{\tilde{a}t}, & a = 1 \\ \theta_{a-1,t} \gamma^s_{a-1} H_{a-1,t} + \omega_t, & a = 2 \\ \theta_{a-1,t} \gamma^b_{a-1} \gamma^s_{a-1} H_{a-1,t}, & a = 3,4,5 \end{cases}, \tag{1}$$

where H_{at} is the number of animals in age cohort (a) during year (t), γ^f is the fraction of calves that are female;[4] θ_{at} is the retention rate (1 – the culling rate); γ^b_a is the birth (calving) rate; γ^s_a is the survival rate (1 – the mortality rate); and ω_t is the number of replacement heifers purchased. The farmer controls the herd size by choosing the retention rates and purchasing replacement heifers, if necessary; the other parameters are fixed. Thus the herd dynamics are characterized by eleven parameters shown in table 1 (γ^f, γ^b_a, $a \in \{1...5\}$ and γ^s_a, $a \in \{1...5\}$), five state variables (H_{at}, $a \in \{1...5\}$) and five control variables (θ_{at}, $a \in \{1...4\}$ and ω_t, with $\theta_{5t} \equiv 0$).

Dairy farmers control their aggregate milk, meat and waste outputs by varying both the herd size and the inputs provided to each cow. In reality, determining the optimal combination of inputs is quite complicated. For example, Rotz et al. (1999a) list thirty different constituents that may be used by farmers to develop a ration. These constituents exhibit fluctuating availabilities (for farm-grown feed), prices (for purchased feed) and qualities, they are marked by complicated patterns of substitutability, and they are bounded by multiple constraints such as the maximum ingestive capacity and the minimum energy requirement of a lactating cow. To simplify this aspect of the problem for our model, we assume each milk cow consumes a fixed cohort-specific ration that contains five common components: alfalfa hay, wheat silage, corn grain, soybean meal and protein mix. Furthermore, because the marginal contributions of each input to milk, meat and waste outputs are largely unknown, we assume that each cow achieves a cohort-specific weight (used to determine the cull price) and produces a fixed amount of milk and waste during each lactation.[5] Table 1 provides these cohort-specific quantities, as well as water requirements and fixed operating costs. With this specification, our herd model exhibits constant returns to scale at the farm level and is consistent with observed trends towards larger dairies. However, as is common for modern dairies, we also include a herd permit constraint that limits the total number of animal units. Table 1 shows the animal unit value for each age cohort.

Given the preceding, we can write the herd component of the profit function as:

$$(2)\ \pi_t^H = p^{milk}\sum_{a=3}^{5}\left[\bar{y}_a H_{at}\right] + \sum_{a=1}^{5}\left[p_a^{herd}\left(1-\theta_{at}\gamma_a^b\right)\gamma_a^s H_{at}\right] + p_1^{herd}\left(\frac{1-\gamma^f}{\gamma^f}\right)H_{1t} - \sum_{a=1}^{5}\left[\mathbf{w}'\mathbf{x}_{at}H_{at}\right] - p^{repl}\omega_t\,,$$

where the first component represents milk sales ($\bar{y}_a$ is per-cow milk yield [kg/yr]); the second represents voluntary culls and sales of cows that fail to calve (p_a^{herd} are the cull prices [\$/cow]); the third represents sales of bull calves; the fourth represents input costs ($\mathbf{w}$ is the input price vector [\$/unit] and $\mathbf{x}_{at}$ is the per-cow input vector); and the fifth represents purchases of replacement heifers (p^{repl} is the price [\$/cow]). Input and output prices are given in table 1.

Waste Management

The second major component of the dairy operation is waste handling and disposal. Because dairy cows are rather inefficient converters of feed into milk (Chang et al. 2005), dairies generate large amounts of both organic and inorganic waste. The amount and nature of the final waste product can vary substantially across dairy farms, depending on the type of housing (e.g., free stall, corral, or open lot), manure collection system (e.g., flush, scrape or vacuum), waste treatment (e.g., solids screening, composting, aerobic or anaerobic digestion), final disposal (e.g., lagoon storage and irrigation, export of dried solids), and environmental conditions (e.g., climate and soil characteristics). In California's Central Valley, where both the dairy industry and nitrogen pollution problems continue to grow, it is common for modern dairies to employ free stall housing with waste flushing, followed by solids screening, lagoon storage and irrigation of liquid waste, and land application or export of separated solids. On such a farm, milk cows spend about 85% of their time in the housing structure and 15% of their time in the milking parlor (Chang et al. 2005). Solid and liquid wastes are deposited in both locations and flushed – with substantial quantities of water – into a solids separator that removes a fraction of the solid content. The liquid waste is stored in an open lagoon and eventually used to irrigate and fertilize crops; the solids are dried, stored, and either sold off the farm or applied as fertilizer. Because this is a typical process for modern dairies and because we have excellent data from a farm like this near Hilmar, California, we specify this type of waste handling system for our model farm.[6]

Even with these specifications, the characteristics of the final waste product depend on numerous decisions made by the farmer, including: the quantity and quality of flush water; the flushing frequency; the amount and type bedding material used; and – because nitrogen is not a conservative pollutant – the residence times in various stages of the waste handling system. To simplify this process for our purposes, we assume that the farmer cannot affect aspects of the waste handling system that occur between waste generation and storage. Rather, for a given quantity of flushed waste (which the farmer affects through herd management decisions), the resulting flows to solid and liquid storage are pre-determined as shown in figure 1. The farmer can then decide how much solid waste to land apply and how much to export; and how much liquid waste to land apply and – by manipulating the size of the lagoon – how much to dispose of through evaporation and volatilization.

We specifically examine the possibility of disposing of liquid dairy wastes via increased volatilization for several reasons. First, although nitrogen emissions to ground water and air historically have been treated as separate problems,[7] each is a result of the same waste stream generated by the milking herd. Therefore, when faced with regulations on emissions into one medium, a dairy farmer naturally would attempt to take advantage of the remaining free disposal option before undertaking costly pollution control measures.[8] It is thus desirable to incorporate such behavior into the model. Second, although there may be other ways to increase ammonia volatilization from a dairy farm that do not involve increasing the size of the storage lagoon, we note that evaporation of saline drainage water is a well-established, cost-effective waste disposal practice for crop producers in the Central Valley. Therefore a similar disposal method seems eminently plausible for dairy farmers when faced with stricter nitrate regulations, particularly farmers using the waste disposal system we have specified for our model. Furthermore, although we can construct a cost curve for lagoon disposal, we do not have data on the marginal costs of manipulating other aspects of the waste disposal system to increase volatilization. To the extent the cost curve for lagoon disposal is representative of other disposal options, our model results apply to dairy farms with other waste handling systems and those that choose other disposal methods. And third, the possibility of disposing of liquid waste via "total evaporation lagoons" has been proposed by extension specialists in both Texas (Harris, Hoffman and Mazac 2001) and New Mexico (Massie 2005) as a means to protect ground water quality in rural areas.

We therefore specify three additional control variables for the waste handling component of our model: e_t is the surface area of evaporation ponds for liquid waste disposal [m^2]; s_{ct} is the amount of solid waste applied to crops [kg N/ha]; l_{ct} is the amount of liquid waste applied to crops [cm]. To simplify the dynamics of our problem we assume no waste is carried-over between crop seasons, implying all waste generated during each season must be land applied or exported off the farm or must volatilize during that season; we assume all inorganic nitrogen is in the ammonia form until it is land applied, at which time the fraction that does not volatilize during application is converted to nitrate (Harter, Mathews and Meyer 2001); and we assume all organic nitrogen is conserved until it is land applied and begins to mineralize. Assuming steady-state conditions in the ponds, the rate of nitrogen flux to the atmosphere (n_t^v) [kg/s] from pond disposal is given by a standard physical relationship (Liang, Westerman and Arogo 2002):

(3) $$n_t^v = K_L F_1 \left[TAN\right] e_t,$$

where K_L [m/s] is the overall mass transfer coefficient for ammonia, F_1 is the fraction of free ammonia concentration in solution, $\left[TAN\right]$ [kg N/m^3] is the total ammonia nitrogen concentration, and e_t [m^2] is defined previously. Table 2 summarizes these and other parameters and relationships used to specify the pond mechanism.

Crop Production

The third and final component of the dairy farm is crop production. Here we follow convention and assume farmers grow two crops annually – summer corn and winter wheat – on a fixed amount of land that is available for either crop production or waste lagoons. Our crop component closely follows that of Schwabe and Knapp (2005) – we borrow empirical relationships for crop yield, nitrogen uptake and nitrogen leaching, and we include non-uniform irrigation – but with two exceptions. First, because dairy wastes include both inorganic and organic species of nitrogen, we have two state variables for soil nitrogen at each field location: $I_{ct}(\beta)$ and $O_{ct}(\beta)$ are the concentrations of inorganic and organic nitrogen at field location β during crop season (c). Second, we include an additional crop (wheat) for which we recalibrate the equations for corn from Schwabe and Knapp (2005) using data from several additional sources (Chang et al. 2005; VanderSchans 2001; Crohn 1996).

Mathematically, the cropping system is expressed as:

(4) $$y_{ct}(\beta) = \bar{y}_c\left(\frac{1}{1+\alpha_1\left(\alpha_2+rf_c+\beta\left(w_{ct}+l_{ct}\right)\right)^{\alpha_3}}\right)\left(\frac{1}{1+\alpha_4\left(n_{ct}^u(\beta)\right)^{\alpha_5}}\right)$$

(5) $$n_{ct}^u(\beta) = \bar{n}_c\left(\frac{1}{1+\alpha_6\left(\alpha_7+rf_c+\beta\left(w_{ct}+l_{ct}\right)\right)^{\alpha_8}}\right)\left(\frac{1}{1+\alpha_9\left(n_{ct}^p(\beta)-n_{ct}^z(\beta)\right)^{\alpha_{10}}}\right)$$

(6) $$n_{ct}^p(\beta) = I_{ct}(\beta)+d_c+f_{ct}+\beta\left(w_{ct}\mu_w+\phi l_{ct}\mu_{lt}^I\right)+\delta_c\left(O_{ct}(\beta)+s_{ct}+\beta l_{ct}\mu_{lt}^O\right)$$

(7) $$n_{ct}^z(\beta) = \frac{\alpha_{11}n_{ct}^p(\beta)}{1+\exp\left(\alpha_{12}\left(rf_c+\beta\left(w_{ct}+l_{ct}\right)+\alpha_{13}\right)\right)}.$$

Equation (4) specifies the yield (y) [T/ha] of crop (c) at time (t) and field location β as a function of the maximum potential yield ($\bar{y}_c$), seasonal rainfall (rf_c) [cm], the parameter β which we discuss below, depth of applied irrigation water (w_{ct}) [cm], depth of applied liquid waste (l_{ct}) [cm], nitrogen uptake rate (n_{ct}^{u}) [kg N/ha], and some parameters α. Equation (5) specifies the nitrogen uptake at field location β as a function of the maximum potential uptake ($\overline{n}_c$), seasonal rainfall, applied irrigation water, applied liquid waste, the amount of plant available nitrogen (n_{ct}^{p}) [kg N/ha], the nitrogen leaching rate (n_{ct}^{z}) [kg N/ha], and some parameters. Equation (6) specifies the amount of plant available nitrogen at field location β as a function of the concentration of inorganic soil nitrogen (I_{ct}) [kg N/ha], the rate of atmospheric nitrogen deposition (d_c) [kg N/ha], the amount of chemical fertilizer applied to crops (f_{ct}) [kg N/ha], the depth of applied irrigation water and its nitrogen concentration (μ_w) [kg N/cm-ha], the depth of applied liquid waste and the associated inorganic nitrogen concentration (μ_{lt}^{I}) [kg N/cm-ha] net of ammonia losses during application (ϕ), the seasonal nitrogen mineralization rate (δ_c), the concentration of organic soil nitrogen (O_{ct}) [kg N/ha], the amount of dried solid waste applied to crops (s_{ct}) [kg N/ha], and the depth of applied liquid waste and the associated organic nitrogen concentration (μ_{lt}^{O}) [kg N/cm-ha].[9] Equation (7) specifies the nitrogen leaching rate at field location β as a function of the amount of plant available nitrogen, seasonal rainfall, depth of applied irrigation water, depth of applied liquid waste and some parameters.

Following Schwabe and Knapp (2005) and citations therein, we interpret the parameter $\beta \in [0, \infty]$ as the water infiltration coefficient and assume it has a log-normal distribution throughout the cropped area with $E[\beta] = 1$ and $\sigma(\beta) = 0.3$. This parameterization corresponds to furrow irrigation with ½-mile runs (UCCC 1988) which is a common type of irrigation system used by dairy farms in our study area. It implies that each point within the cropped area receives a positive fraction of the average applied water depth for the entire field and thus provides a model for non-uniform irrigation. To make this model tractable, we discretize the support for β into five intervals and treat these intervals as distinct field location types, each with its own specific water infiltration coefficient $\beta_j,\ j \in \{1...5\}$. This discretization allows us to specify the seasonal equations of motion for the soil nitrogen concentrations at each field location β. Assuming wheat (c') follows corn (c) during each year (t), these equations are given by:

(8) $$I_{c't}(\beta) = (1-\lambda_c)\,n_{ct}^{p}(\beta) - n_{ct}^{z}(\beta) - n_{ct}^{u}(\beta),$$

(9) $$I_{c,t+1}(\beta) = (1-\lambda_c)\,n_{c't}^{p}(\beta) - n_{c't}^{z}(\beta) - n_{c't}^{u}(\beta),$$

(10) $$O_{c't}(\beta) = (1-\delta_c)\left(O_{ct}(\beta) + s_{ct} + \beta l_{ct}\mu_{lt}^{O}\right), \text{ and}$$

(11) $$O_{c,t+1}(\beta) = (1-\delta_{c'})\left(O_{c't}(\beta) + s_{c't} + \beta l_{c't}\mu_{lt}^{O}\right),$$

where λ_c accounts for seasonal losses due to denitrification and all other variables have been defined previously.

Given the preceding, we can write the crop component of the profit function as:

(12)

$$\pi_t^C = \sum_c \begin{pmatrix} p_c^{crop}\sum_j y_{ct}(\beta_j)F(\beta_j) - p^{sol}s_{ct} - w_c^{fix} \\ -w^i i_{ct} - w^f f_{ct} - w^z \sum_j n_{ct}^{z}(\beta_j)F(\beta_j) \end{pmatrix}\left(L - \frac{e_t}{10^4}\right) + p^{sol}\overline{S}_t - w^v\left(n_t^v + n_t^{hs}\right) - w^e \max_{\tau\in\{1...t\}}\left[e_\tau\right],$$

where p_c^{crop} is the price received for crop (c) [\$/T]; $F(\beta_j)$ is the probability that β is in the interval corresponding to β_j; p^{sol} is the price received for exported solid waste [\$/kg N]; w_c^{fix} is the fixed production cost [\$/ha]; w^i is the cost of irrigation water [\$/cm-ha] and i_{ct} is the amount of irrigation water applied [cm]; w^f is the cost of chemical fertilizer [\$/kg N] and f_{ct} is the amount of chemical fertilizer applied [kg N/ha]; w^z is the cost of nitrate leaching to ground water [$/kg N];[10] L is the total amount of land available for crops and ponds; $\overline{S}_t$ is the total amount of dried solid waste going to storage [kg N] (see figure 1); w^v is the cost of ammonia volatilization to the atmosphere [$/kg N]; n_t^{hs} is the amount of ammonia volatilization from housing and storage [kg N] (see figure 1); w^e is the annualized cost of constructing evaporation ponds [$/m^2]; and the other variables have been defined previously. The max function specifies that a farmer must continue to pay the annualized cost for the maximum constructed pond area to-date even if less than the total area is used in the future (e.g., if the herd size is reduced). Table 3 summarizes the parameter values for the cropping system.

Optimization

Defining $\pi_t \equiv \pi_t^H + \pi_t^C$, collecting all prices into a vector $\mathbf{p}$ and all parameters into a vector $\boldsymbol{\Gamma}$, specifying a discount factor ρ and a time horizon T, and assuming farmers maximize the net present value of farm operations, we can summarize the producer's problem as:

$$\max_{\{\theta_{at},\omega_t,e_t,s_{ct},l_{ct},f_{ct},i_{ct}\}} \left[\sum_{t=0}^{T} \rho^t \pi_t \left(H_{at}, I_{ct}^{\beta}, O_{ct}^{\beta}, \theta_{at}, \omega_t, e_t, s_{ct}, l_{ct}, f_{ct}, i_{ct} \mid \mathbf{p}, \boldsymbol{\Gamma} \right) \right], \tag{13}$$

subject to equations (1) - (12), constraints on total land and total animal units, mass balance constraints on solid and liquid waste streams, and a requirement that liquid waste must be sufficiently diluted with irrigation water before applying to crops in order to avoid damaging the plants with high concentrations of waste components that do not volatilize (e.g., salts) and therefore become concentrated in the residual pond water (Swenson 2004). Equations (1) - (13) and the associated constraints define an optimal control problem with fifteen state variables (five herd age cohorts and ten soil nitrogen concentrations – inorganic and organic nitrogen at each of five locations β) and fifteen control variables (five culling rates, the number of purchased replacement heifers, the evaporation pond area, and two crop-specific values for applications of solid waste, liquid waste, chemical fertilizer, and irrigation water). We solve this dynamic optimization problem in GAMS as a constrained non-linear programming problem (Standiford and Howitt 1992) and utilizing the CONOPT solver.[11] With this approach, the state equations (1) and (8) - (11) are treated as constraints that apply to each time step in the model. Our first goal is to find a dynamic steady state and verify that our model farm is representative of our study site in Hilmar, California; then we conduct policy simulations. To find feasible starting values for our steady-state search, we first treat the model as a period-by-period optimization problem: we choose a set of initial conditions, optimize the first period in isolation from the others, use the state equations to "roll forward" to the next period, and continue until the last period (which is set arbitrarily large to avoid any boundary effects). We then solve the dynamic problem using the period-by-period solution as the starting values, check to see if the model farm has reached a steady-state, select a new set of initial conditions from the dynamic solution path, and repeat until steady-state convergence criteria are satisfied.

Model Calibration Results

Table 4 summarizes the results of our model calibration by comparing various steady state values against available data. Despite the large number of parameters, variables, and equations, and the complexity of the optimization problem, our model farm appears to be calibrated rather well. Animal numbers are similar to those reported by VanderSchans (2001) for the Hilmar site. Differences are most likely due to off-farm rearing of calves and heifers (a strategy which is not chosen by our model farm). Income data is not available for the Hilmar farm, but we can compare our annual profit per cow against Rotz et al. (2003) who simulate a 1,000 cow dairy with 770 heifers and 600 hectares of cropland. Our profit of $784/cow is low compared to their estimate of $1,309/cow, but this appears to be due to different assumptions about milk yield.

The average annual milk yield for our herd is 9,509 kg/cow whereas the average for the simulation in Rotz et al. (2003) is 11,300 kg/cow. Substituting 11,300 kg/cow into our model gives annual profit of $1,392/cow, which is very close to their estimate. However, in 2004 the statewide average milk yield for California dairies was 9,494 kg/cow (USDA 2006); therefore we do not use the higher yield. Ammonia volatilization from our model farm is similar to reported values (which vary widely), and nitrate leaching is very close to VanderSchans' best estimate for the Hilmar farm. Corn and wheat yields are high but within reason, and concentrations of nitrogen in the manure storage lagoon are within acceptable ranges. Applied water (irrigation plus lagoon water) is close to the Hilmar farm estimate, but applied chemical fertilizer is significantly different. Our model farm does not apply any chemical fertilizer, which supports claims by Chang et al. (2005) that California dairies can achieve high crop yields without chemical fertilizers; but it contradicts observed practice at the Hilmar site which involves the application of 130-280 kg N/ha-yr. However, the only noteworthy change derived from imposing the midpoint application rate of 205 kg N/ha-yr on our model is an increase in the leaching rate from 392 kg N/ha-yr to 461 kg N/ha-yr (which remains close to VanderSchans' best estimate of 417 kg N/ha-yr for the Hilmar farm). Lastly, our model farm sells and exports all dried solid manure off-farm, which is consistent with the discussion in VanderSchans (2001).

Policy Simulations

The calibration results suggest that, given a set of economic conditions, our model is capable of generating a realistic operating position for a modern dairy. Changing any of the economic conditions will change the operating position, but it is not obvious how long it will take the dairy to transition from one steady state to another or what will be the properties of the new steady state. Answers to both of these questions – the nature of the transition period and the eventual steady state operating position – are needed to evaluate policies for addressing the dairy nitrogen problem. Policies that generate long transition times, those that result in undesirable steady state levels, and those that impose substantial costs on farmers are unlikely to be successful.

Policy mechanisms that have received considerable attention in the literature include best management practices (i.e., technology incentives and standards), nutrient management plans (i.e., waste disposal standards), water input taxes, nitrogen input taxes, and nitrogen emission taxes. Here we examine policies that effectively increase the cost of nitrogen disposal. Such policies could take a variety of forms when implemented; we model these by introducing prices for nitrogen emissions to ground water and air. We examine emissions taxes in particular because they are straightforward to model and can achieve any desired level of nitrogen emissions cost-effectively. Our model could incorporate other types of policy mechanisms, but we leave these for future work and instead focus on an instrument that will produce general insights into the likely response of dairy farms to stricter nitrogen regulations. However, we recognize that nitrogen emissions are diffuse and therefore the problem of estimating emissions must be overcome before emissions taxes could be implemented. Therefore we implicitly assume the regulator possesses a model of dairy nitrogen emissions similar to ours that is used by both parties (regulator and farmer) to determine emissions levels.

Our policy simulations assume the dairy farm is initially at the steady state operating position derived in the model calibration section. We then introduce positive prices for nitrogen emissions to ground water and/or air and we derive the dynamically optimal transition path for the dairy. We focus our attention on the time paths for three variables: herd size [number of milk cows], nitrate leaching [kg N/ha-yr], and ammonia volatilization [kg N/yr], as well as the net present value (NPV) of farm operations during the simulated time period (60 years). We conduct our policy simulations using combinations of $w^z \in [0,50]$ and $w^v \in [0,15]$.

Figure 2 presents the NPV of farm operations for different emissions tax vectors as a percentage of the NPV that is attainable without taxes. This NPV can be considered a rough estimate of the market value of the dairy farm assuming 60 years is a sufficiently long time horizon. Without taxes the baseline NPV of farm operations is $25.0 million; but figure 2 shows that this value erodes quickly as the emissions taxes are increased. With taxes of $5/kg N for leaching and $2.50/kg N for volatilization, the NPV is only $16.6 million (66% of the baseline). Raising the cost of nitrogen disposal will create a significant financial burden for modern dairies.

Figures 3 – 5 summarize the tax-induced steady-state values for the other variables of concern. Figure 3 shows the new steady state herd size will continue to be large except for relatively high leaching and/or volatilization taxes that lead to substantially lower profits. Figures 4 and 5 show nitrate leaching and ammonia volatilization also remain near their baseline steady-state values even for moderate tax levels. A leaching tax of $5/kg N and a volatilization tax of $2.50/kg N will induce a new farm steady state with 1410 milk cows, 391 kg N/ha leached and 81,672 kg N volatilized each year. These numbers are nearly identical to the baseline steady-state values of 1413 milk cows, 391 kg N/ha leached and 81,679 kg N volatilized each year. This tax policy would produce sizeable revenues (approximately $375,000 annually), but only by extracting a considerable financial loss from the dairy farmer ($8.4 million as NPV). These results suggest that the operating positions of modern dairies are relatively inelastic – which is consistent with non-decreasing returns to scale in production and the lack of a viable substitute for nitrogen– and that financial incentives must be quite large to stimulate significant changes in production and waste generation.[12]

Figures 4 and 5 also demonstrate the likely outcome of regulating only nitrate leaching or ammonia volatilization. A leaching tax of $20/kg N would cut nitrate leaching by about 50%, but it also would induce a 60% increase in ammonia volatilization if volatilization remains unregulated. This cross-media pollution problem is the result of (1) the strong incentive –due to economies of scale and lack of substitutes– to maintain a large herd and a high rate of nitrogen throughput; (2) mass balance constraints; and (3) the lack of an economical treatment method for converting waste nitrogen into a benign byproduct. It is apparent that candidate solutions to water and air pollution problems in dairy-intensive areas should be evaluated concurrently.

Figures 6 – 8 show the optimal time paths for herd size, nitrate leaching, and ammonia volatilization for five relatively moderate tax vectors $\left[w^z, w^v\right]$. The most interesting results are for the [10,10], [20,5], and [20,0] policies. The [20,0] policy does not change the herd size significantly, but it leads to a gradual 49% reduction in leaching over approximately 5 years at the expense of a rapid 62% increase in ammonia volatilization. The leaching curve is smooth because a large amount of water must still be applied to fields in order to sufficiently dilute other conservative components of the waste stream that can damage crops. Because the nitrogen concentration of this water is lower than it was during the pre-tax steady-state, both the stock of soil nitrogen and the amount of leached nitrogen are gradually reduced. In contrast, the [20,5] and [10,10] policies lead to relatively fast reductions in all variables. The leaching rate drops precipitously because the smaller herd size means less water must be applied to fields to achieve adequate dilution. For all three policies, the new steady state is achieved within about eight to nine years. Despite the additional elements of livestock-crop operations that tend to delay the response of producers to changing operating conditions, it is apparent that the transition time to a new steady-state is approximately the same as for crop-only operations (Schwabe and Knapp

2005). However, we suspect that the transition time could be significantly longer for livestock operations involving longer animal residence times.

A final set of policy simulations examines the implications of announcing the tax policy at t=0, but delaying its implementation for at least one year. This approach has the potential to alleviate some of the financial burden on the farmer by providing an adjustment period during which waste disposal remains free; but it creates the opportunity for a farmer to flush a large portion of the soil nitrogen stock into ground water without penalty. Figure 9 shows the effect on nitrate leaching of delaying the implementation of the [20,5] tax policy by 1, 2, or 3 years. As expected, farmers take advantage of the grace period to dispose of their stocks of soil nitrogen by applying excess water and flushing additional nitrates to ground water. The effect of such an approach on the NPV of farm operations is not insignificant, but NPV remains low. With no grace period, NPV = $7.99 million; with a 1-year grace period, NPV = $8.97 million; with a 2-year grace period, NPV = $9.68 million; and with a 3-year grace period, NPV = $10.36 million. Delaying policy implementation clearly benefits farmers, but the cost of temporarily higher ground water nitrate concentrations must be weighed against this.

It is important to emphasize that all of the results presented here should be interpreted carefully. If the cost of nitrogen disposal were to increase substantially, other currently unproven or uneconomical waste treatment technologies could become viable options. For example, small-scale nitrification/denitrification (N/D) systems might be used to reduce both nitrate leaching and ammonia volatilization significantly without reducing herd sizes. N/D systems are used by large-scale municipal waste water treatment plants to convert aqueous ammonia to molecular nitrogen gas, an inert compound which comprises 78% of the atmosphere. Small-scale N/D systems have been tested and shown to be effective in experimental settings, but both performance and cost data remain very sketchy (SJVDMTFAP 2005). Estimated installation, operating and maintenance costs are high – on the order of hundreds of dollars per cow annually; and the system may be too complex for a dairy farmer to operate effectively without professional assistance (SJVDMTFAP 2005). Furthermore, N/D systems entail an additional cost for dairy farmers because they convert waste nitrogen into a form that has no value as fertilizer. And it is likely that effective treatment would require modifying the animal confinement structures in order to capture ammonia emissions that occur before the waste stream reaches the N/D system. However, as better performance and cost data become available, technologies like these can be incorporated into whole-farm models like ours to provide more precise estimates of the profit and emissions functions for relatively high nitrogen disposal costs.

Conclusion

Simulation models of dairy farms have been used in the animal science and agricultural economics literatures to predict the outcomes of various farm management strategies. Previous efforts have focused on optimizing animal replacement or other farm management decisions without considering environmental impacts (e.g., Tozer and Huffaker 1999; Chavas and Klemme 1986; Van Arendonk 1985). Others have modeled the entire farm, including waste generation, but without an economic optimization component (i.e., pure simulation models as in Rotz et al. 2003, 1999a, 1999b). Schnitkey and Miranda 1993 is a notable exception, although the application is different and the model is not as comprehensive as ours.

In this article we develop a very detailed micro-dynamic model of a modern dairy farm, including milk and livestock production, waste generation, treatment, and disposal, and crop production with non-uniform irrigation. The model is calibrated with farm-level data from a well-documented dairy in California's Central Valley and with additional data from published sources. The optimized characteristics of the farm, including herd size, crop yield, amounts of applied water and fertilizer, nitrate leaching, ammonia volatilization, and net farm income, are consistent with available comparison data and suggest that the model can be used to reliably predict policy outcomes.

Simulations of policies that increase the cost of nitrate and ammonia disposal demonstrate three main results: (1) dairies are unresponsive to pollution charges unless they are relatively large and thus financially burdensome for farmers; (2) regulations aimed at controlling only nitrate leaching will cause significant increases in ammonia emissions; and (3) mitigating both nitrogen problems with emission taxes involves substantial reductions in both herd size and farm profit. We also find that dynamic optimizing behavior leads to management response delays that are of the same magnitude as for crop-only operations, as well as to excess dumping of nitrogen stocks to ground water during the run-up to policy implementation.

Our results collectively suggest that marginal cost pricing of environmental services is not, by itself, a practical solution to the dairy nitrogen problem. Financial responsibility most likely will have to be shared with other stakeholders. Subsidizing on-farm waste treatment technologies may be necessary to reduce environmental impacts while maintaining the existing structure of the industry (i.e., high stocking densities), but additional research is needed to better characterize such technologies. The economics of pumping and treating nitrate contaminated ground water prior to human use also should be examined. And regional cost-benefit analyses of the dairy industry, its environmental and economic impacts, and the agricultural policies that affect it should be considered where pollution problems are particularly acute.

Endnotes

1 Other studies (e.g., Nkonya and Featherstone 2000; Yadav 1997; Kim, Hostetler and Amacher 1993) have demonstrated the importance of dynamic elements affecting the fate and transport of nitrates in the environment. As with the studies cited in the main text, this article focuses on dynamic elements of the production process. Linking a model like ours to a dynamic fate and transport model would allow a more complete analysis of the problem.

2 Ammonia reacts with oxides of nitrogen and sulfur to form PM2.5 (and PM10). Currently the San Joaquin Valley Air District is in non-attainment status for both of these criteria air pollutants (SJVAPCD 2006). In 2000, approximately 43% of the ammonia emissions in the San Joaquin Valley were from dairy farms (Palsgaard 2006).

3 The livestock management literature contains studies (e.g., Van Arendonk 1985) that examine the optimal time to cull individual animals. In reality, these are important decisions that are based on animal-specific characteristics. To make the herd management problem tractable for our analysis, we omit these characteristics and impose a maximum three-year residence time for each cow in the milking herd (Tozer and Huffaker 2000). This means each animal spends at most five years on the dairy farm, which is typical for modern dairies.

4 We assume bull calves are sold during their first year.

5 Although some dairies, particularly smaller and older ones, choose relatively low to moderate levels of per-cow milk output, most modern operations consistently aim for very high per-cow milk output levels. Our output specification is consistent with this practice.

6 In the log run all aspects of production and waste management are variable. But such an optimization problem typically is intractable and economists regularly hold certain factors fixed in order to examine the role of other variables that are thought to be of key importance.

Given the already large number of variables in our model and the main objective of this research, we leave an investigation of these waste management decisions for future work.

7 For example, many of the manure management strategies suggested by the Dairy Permitting Advisory Group for the San Joaquin Valley Air Pollution Control District involve shifting emissions from ammonia to nitrate (Abernathy et al. 2006).

8 The National Research Council (2002) has emphasized this and that new water quality regulations could have either positive or negative impacts on air emissions from CAFOs.

9 We assume dried solid waste has negligible inorganic nitrogen due to the drying and storage process during which large amounts of ammonia-nitrogen volatilize (Chang et al. 2005).

10 Nitrates also leach from lagoons and feedlots. Generally these leaching rates are much less significant due to low permeability liners, compacted soils, and small land areas. Crop field leaching typically accounts for about 80% of total leaching (Campbell Mathews 2006). Here we assume the lagoon has an effective liner and cows spend little time on unpaved surfaces.

11 The advantage of using GAMS is that it can readily solve high-dimensional dynamic optimization problems, but it cannot easily incorporate stochastic state equations. Incorporating stochastic state equations (e.g., for prices, technologies, etc.) would require a different solution method such as stochastic-dynamic programming, but this is not well-suited for high-dimensional problems. Our deterministic problem framework establishes a baseline from which future investigations into the role of uncertainty can be conducted.

12 For comparison, Schwabe and Knapp (2005) estimate that a leaching tax of $1/kg N achieves a 58% reduction in nitrate leaching (from 36 kg N/ha to 15.2 kg N/ha) and reduces net farm income by 13% for a similar crop-only enterprise.

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Table 1. Herd Production Parameters.

Symbol	Description	Value	Source
γ^f	Fraction of newborn calves that are female	0.5	Tozer and Huffaker 2000
γ_a^s	Fraction of each age cohort that survives	0.95	Tozer and Huffaker 2000
γ_a^b	Fraction of survivors from cohort (a) that calve	{0, 0.96, 0.96, 0.96, 0}	Tozer and Huffaker 2000
x_{a1}	Alfalfa hay consumed by cohort (a) [kg/cow-yr]	{270, 690, 861, 861, 861}	Rotz et al. 1999a; Chang et al. 2005
x_{a2}	Wheat silage consumed by cohort (a) [kg/cow-yr]	{861, 2143, 2621, 2621, 2621}	Rotz et al. 1999a; Weiss et al. 1995; Chang et al. 2005
x_{a3}	Corn grain consumed by cohort (a) [kg/cow-yr]	{522, 102, 3296, 3296, 3296}	Rotz et al. 1999a; Weiss et al. 1995; Chang et al. 2005
x_{a4}	Soybean meal consumed by cohort (a) [kg/cow-yr]	{0, 0, 13, 13, 13}	Rotz et al. 1999a; Chang et al. 2005
x_{a5}	Protein mix consumed by cohort (a) [kg/cow-yr]	{0, 0, 151, 151, 151}	Rotz et al. 1999a; Chang et al. 2005
x_{a6}	Water – for drinking, washing, cleaning, flushing and cooling – required by cohort (a) [m^3/cow-yr]	{1.42, 3.37, 235.55, 237.25, 237.25}	Murphy et al. 1983; Holter and Urban 1992; Waldner and Looper 2004; van Horn et al. 2003
$\bar{y}_a$	Milk produced by cohort (a) [kg/cow-yr]	{0, 0, 8386, 10270, 10270}	Rotz et al. 1999a

--	Urine produced by cohort (a) [m^3/cow-yr]	{3.81, 5.22, 6.36, 6.36, 6.36}	Wilkerson et al. 1997
--	Urinary N produced by cohort (a) [kg/cow-yr]	{17.64, 47.88, 74.34, 74.34, 74.34}	Wilkerson et al. 1997; Dou et al. 1996; Chang et al. 2005
--	Fecal N produced by cohort (a) [kg/cow-yr]	{13.86, 36.54, 69.30, 69.30, 69.30}	Wilkerson et al. 1997; Dou et al. 1996; Chang et al. 2005
p^{milk}	Price received for milk [$/kg]	0.310	USDA 2006
p_a^{herd}	Price received for culling cohort (a) [$/animal]	{353, 838, 633, 633, 633}	USDA 2006; Wattiaux 1999; Chang et al. 2005
p^{repl}	Price paid for replacement heifers [$/cow]	1500	USDA 2006
$\{w_1 ... w_5\}$	Prices paid for feed components [$/kg]	{0.1624, 0.1432, 0.1444, 0.3008, 0.3971}	Rotz et al. 1999b; Vargas et al. 2003; Brittan et al. 2004
w_6	Price paid for water [$/m^3$]	0.0258	Vargas et al. 2003
w_a^{fix}	Fixed production cost for cohort (a) [$/cow]	{0, 0, 1309, 1309, 1309}	Rotz et al. 2003
--	Animal unit value for cohort (a)	{0.32, 0.73, 0.98, 0.98, 0.98}	CRWQCB 2001
--	Maximum allowable animal units (herd permit)	2069	VanderSchans 2001

Note: all prices are expressed in 2005 dollars using the US Bureau of Labor Statistics producer price index for farm products.

Table 2. Manure Storage Lagoon Specification.

Symbol	Description	Value	Source
$e_t^{\min}$	Minimum pond area [m^2]	11,000	VanderSchans 2001
$e_t^{\max}$	Maximum pond area [m^2]	See note (a)	--
$temp$	Pond temperature [°K]	298	Chang et al. 2005
$wind$	Wind velocity [m/s]	3.13	WRCC 2006
pH	Pond pH level	7.6	VanderSchans 2001
--	Evaporation rate [m/s]	5.60×10^{-8}	WRCC 2006
H_N	Henry's Law constant	$\frac{2.395\times10^5}{temp}\exp\left(\frac{-4151}{temp}\right)$	Liang et al. 2002
k_G	Gas-phase mass transfer coefficient [m/s]	$5.317\times10^{-5}+2.012\times10^{-3}\cdot wind$	Liang et al. 2002
k_L	Liquid-phase mass transfer coefficient [m/s]	$2.229\times10^{-6}\exp(0.236\cdot wind)$	Liang et al. 2002
K_L	Overall mass transfer coefficient [m/s]	$\frac{H_N k_G k_L}{H_N k_G + k_L}$	Liang et al. 2002
K_d	Dissociation constant	$5.2\times10^{-\left(1.0897+\frac{2729}{temp}\right)}$	Liang et al. 2002
F_1	Fraction of free ammonia	$\frac{K_d}{K_d+10^{-pH}}$	Liang et al. 2002

[a] The maximum pond area [m^2] is calculated by dividing the liquid waste generation rate [m^3/s] by the evaporation rate [m/s]. We use 90% of this value to avoid possible division by zero while calculating the pond concentration during optimization and because components of the liquid waste stream that do not volatilize must still be flushed from the lagoon.

Table 3. Crop Production Parameters.

Symbol	Description	Value	Source
rf_c	Rainfall during growing seasons for corn, wheat [cm]	{10, 25}	WRCC 2006
$\bar{y}_c$	Maximum potential yield for corn, wheat [T/ha]	{12.085, 10}	Schwabe and Knapp 2005; Crohn 1996
α_1	Crop yield parameter	103813.82	Schwabe and Knapp 2005
α_2	Crop yield parameter	25	Schwabe and Knapp 2005
α_3	Crop yield parameter	-3.3963	Schwabe and Knapp 2005
α_4	Crop yield parameter	3221.36	Schwabe and Knapp 2005
α_5	Crop yield parameter	-1.812	Schwabe and Knapp 2005
$\bar{n}_c$	Maximum potential nitrogen uptake for corn, wheat [kg N/ha]	{351.87, 250}	Schwabe and Knapp 2005; Crohn 1996
α_6	Nitrogen uptake parameter	58.977	Schwabe and Knapp 2005
α_7	Nitrogen uptake parameter	25	Schwabe and Knapp 2005
α_8	Nitrogen uptake parameter	-1.311	Schwabe and Knapp 2005
α_9	Nitrogen uptake parameter	46926.37	Schwabe and Knapp 2005

α_{10}	Nitrogen uptake parameter	-2.034	Schwabe and Knapp 2005
α_{11}	Nitrogen leaching parameter	0.144	Schwabe and Knapp 2005
α_{12}	Nitrogen leaching parameter	-0.238	Schwabe and Knapp 2005
α_{13}	Nitrogen leaching parameter	-71.41	Schwabe and Knapp 2005
d_c	Atmospheric nitrogen deposition during growing seasons for corn, wheat [kg N/ha]	{4.67, 3.33}	VanderSchans 2001
μ_w	Nitrogen concentration of irrigation water [kg N/cm-ha]	0.1	VanderSchans 2001
ϕ	Fraction of applied liquid waste nitrogen (ammonia) that does not volatilize during application	0.75	Chang et al. 2005
δ_c	Seasonal nitrogen mineralization rate for corn, wheat	{0.473, 0.204}	Chang et al. 2005
λ_c	Seasonal denitrification rate for corn, wheat	{0.25, 0.25}	Meisinger and Randall 1991
p_c^{crop}	Price received for corn, wheat [$/T]	{117, 116}	Vargas et al. 2003; Brittan et al. 2004
p^{sol}	Price received for dried solid manure [$/kg N]	0.14	Norwood, Luter and Massey 2005; Vargas et al. 2003
w_c^{fix}	Fixed crop production cost for corn, wheat [$/ha]	{1524, 629}	Vargas et al. 2003; Brittan et al. 2004

w^i	Price paid for irrigation water [$/cm-ha]	2.58	Vargas et al. 2003
w^f	Price paid to apply chemical nitrogen fertilizer [$/kg N]	0.59	Schwabe and Knapp 2005
L	Total land available for crops and lagoons [ha]	88.62	VanderSchans 2001
w^e	Annualized cost to increase pond size [$/m^2]	0.44[a]	Moser et al. 1998
ρ	Discount factor	0.9615[b]	Schwabe and Knapp 2005

Note: all prices are expressed in 2005 dollars using the US Bureau of Labor Statistics producer price index for farm products.

[a] Assuming a HDPE-lined pond with depth of 1 meter.

[b] Corresponds to a discount rate of 4%.

Table 4. Model Calibration Results.

Quantity	Units	Steady State Value	Comparison Value	Comparison Source
Calves	# of animals	707	517	VanderSchans 2001
Heifers	# of animals	627	308	VanderSchans 2001
Milk cows	# of animals	1413	1731	VanderSchans 2001
Replacement heifers purchased	# of animals	0	--	--
Annualized profit per milk cow ($2005)	$/head	784[a]	1309	Rotz et al. 2003
Ammonia volatilization	kg N/head-yr	40[b]	38	USEPA 2004
			64	Chang et al. 2005
Nitrate leaching	kg N/ha-yr	392	417	VanderSchans 2001
Corn yield	T/ha-yr	10.8	6.7-13.3	Vargas et al. 2003
			7.2-10.0	Crohn 1996
Wheat yield	T/ha-yr	7.9	4.2-6.7	Brittan et al. 2004
			2.7-7.7	Crohn 1996
Lagoon nitrogen concentration [TAN]	mg N/l	910	200-1000	VanderSchans 2001
			500-800	Campbell Mathews 2006
Lagoon inorganic nitrogen concentration	mg N/l	401	300-600	Chang et al. 2005
Applied water (irrigation + pond)	cm/yr	109	124	VanderSchans 2001

Applied chemical fertilizer	kg N/ha-yr	0[c]	130-280	VanderSchans 2001
Applied solid manure	kg N/ha-yr	0[d]	--	--

[a] Comparison value is for a 1000 cow dairy with 770 heifers and 600 hectares of cropland. Difference in annual profit is due to milk yield per cow. Rotz et al. (2003) assume 11,300 kg/yr for all lactations. Imposing this assumption on our model gives annual profit per cow of $1392. We retain the milk yield parameters reported in table 1 because they are much closer to the reported average for California dairies (USDA 2006) and because we do not have milk yield and profit data for the Hilmar site.

[b] Includes heifers and milk cows but not calves. Annual volatilization per milk cow is 58 kg N.

[c] Imposing the midpoint of VanderSchans' range (205 kg N/ha-yr) on our model gives a leaching rate of 461 kg N/ha-yr.

[d] VanderSchans suggests solid manure generated by the Hilmar farm typically is sold for off-farm application.

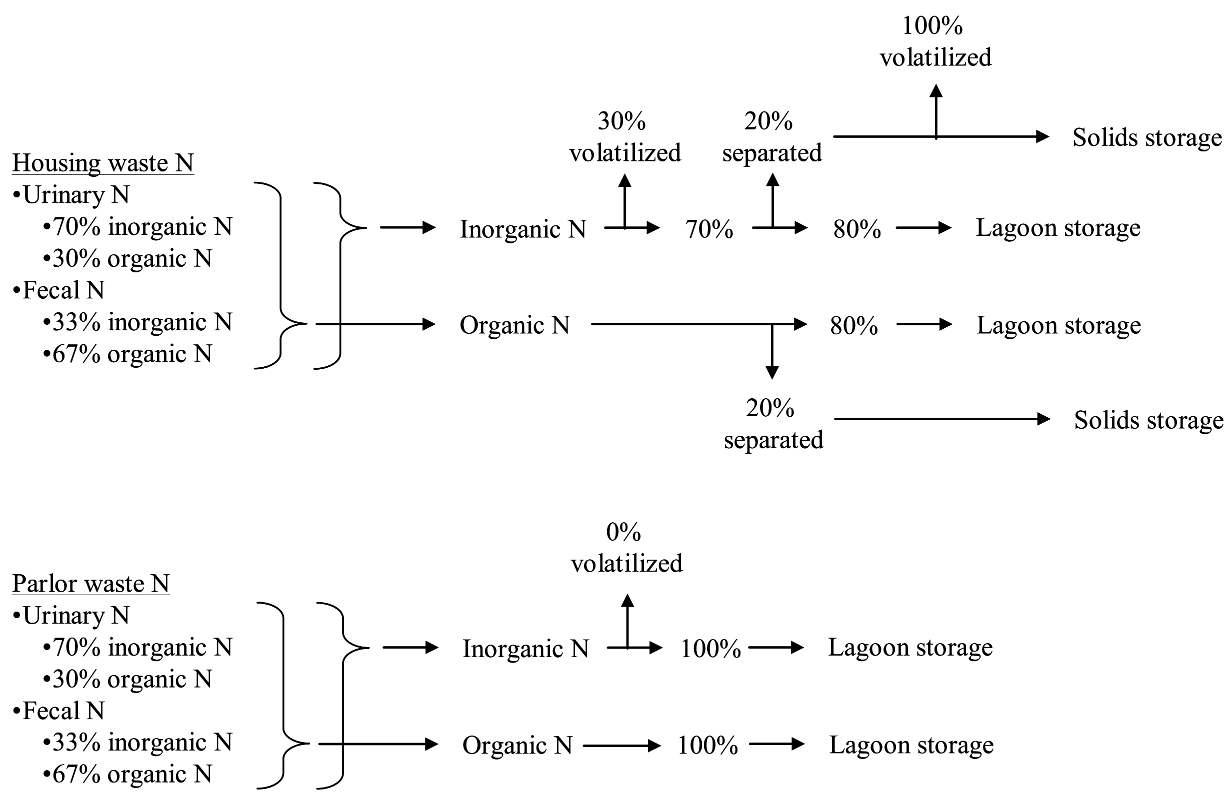

Figure 1: Predetermined waste nitrogen flows (Chang et al. 2005; VanderSchans 2001).

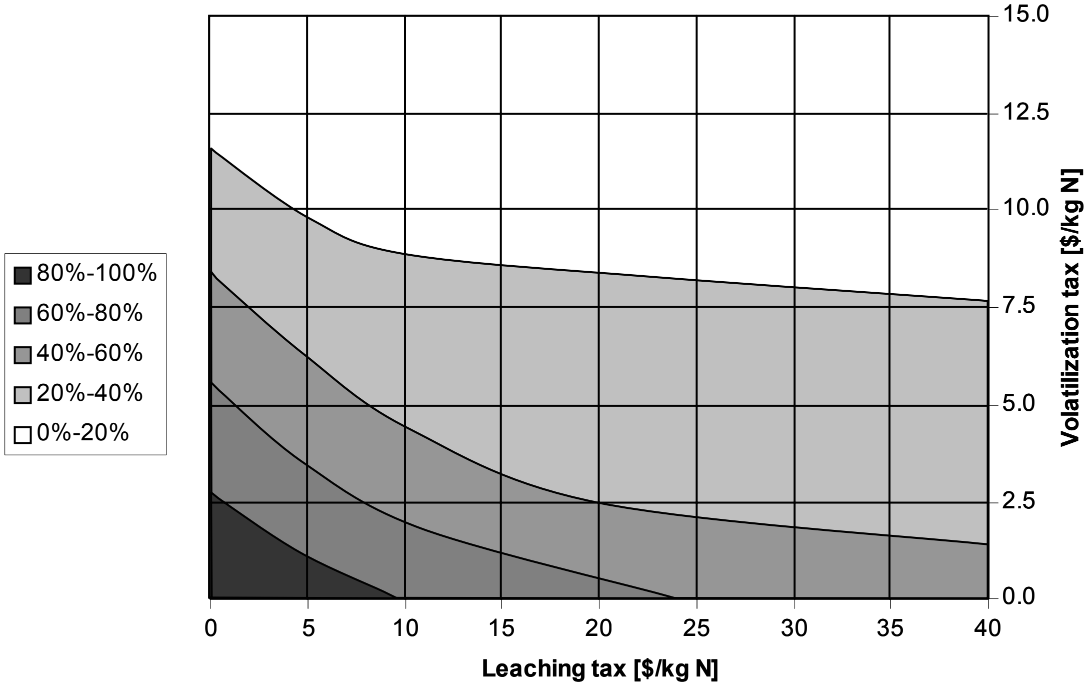

Figure 2: Net present value of farm operations with emissions taxes as a percent of net present value without taxes. Contours are interpolated.

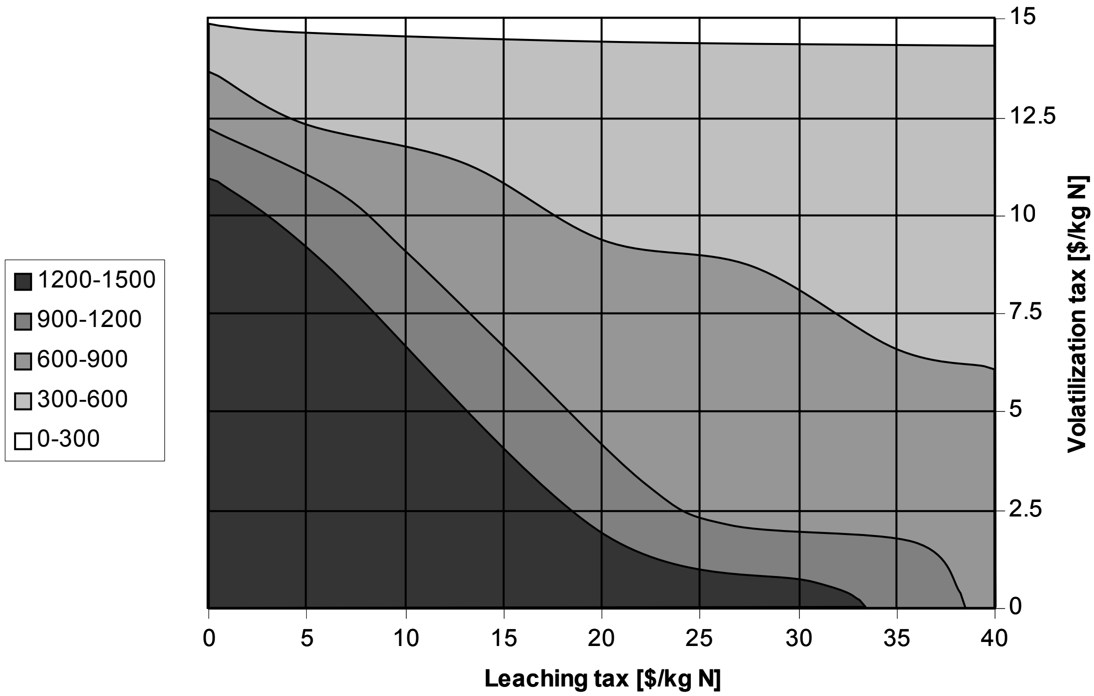

Figure 3: Steady-state herd size [number of milk cows] for different emissions tax vectors. Contours are interpolated.

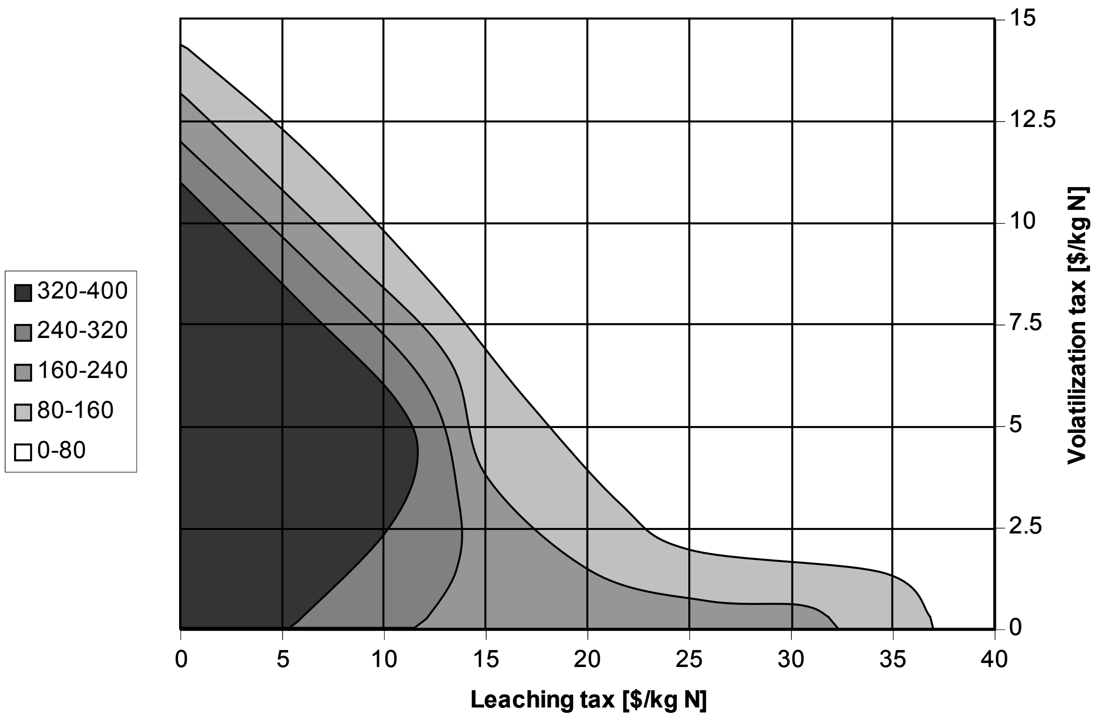

Figure 4: Steady-state leaching rate [kg N/ha-yr] for different emissions tax vectors. Contours are interpolated.

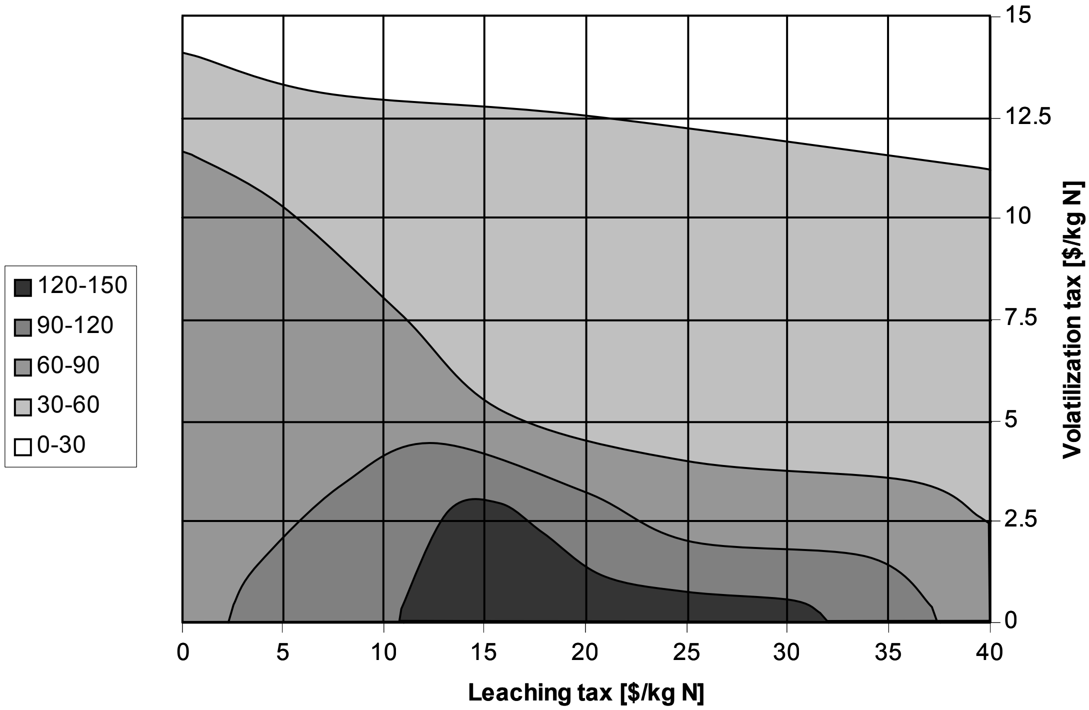

Figure 5: Steady-state volatilization rate [tonnes N/yr] for different emissions tax vectors. Contours are interpolated.

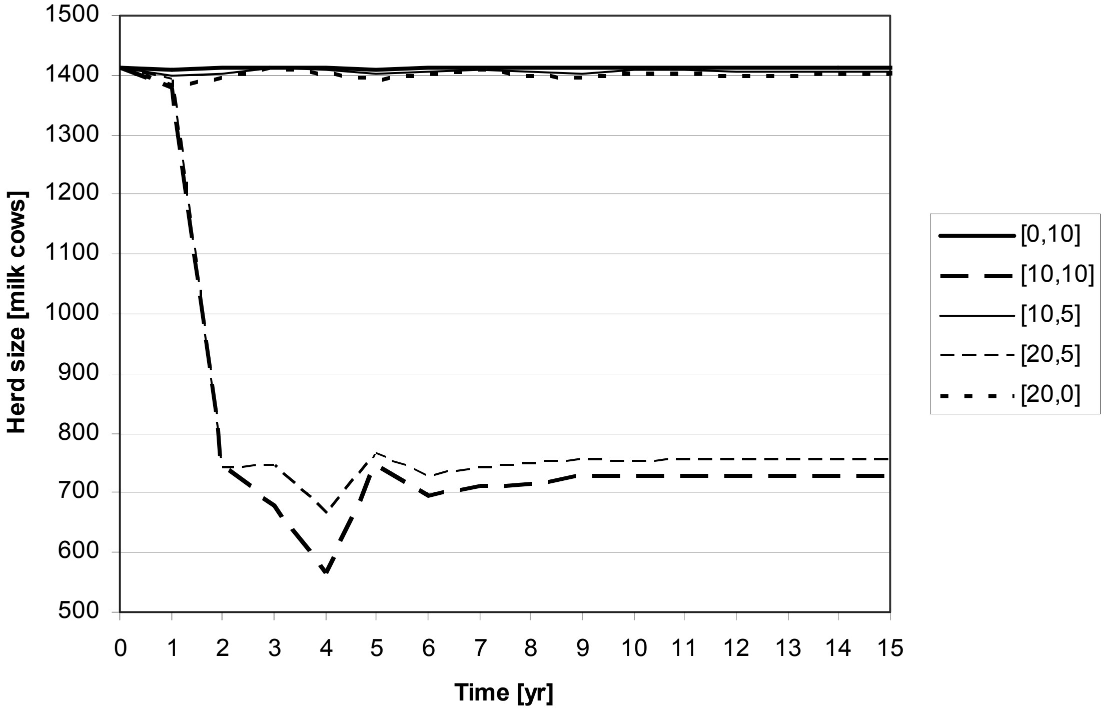

Figure 6: Time path of herd size for different emissions tax vectors $\left[w^z, w^v\right]$.

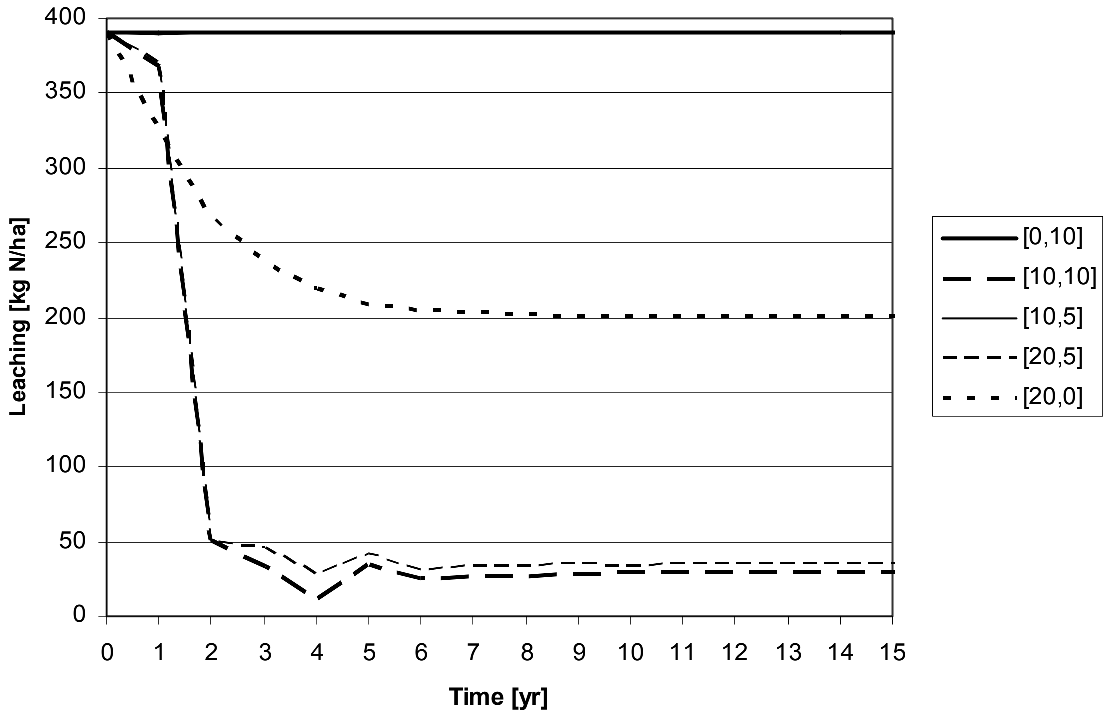

Figure 7: Time path of nitrate leaching for different emissions tax vectors $\left[w^z, w^v\right]$.

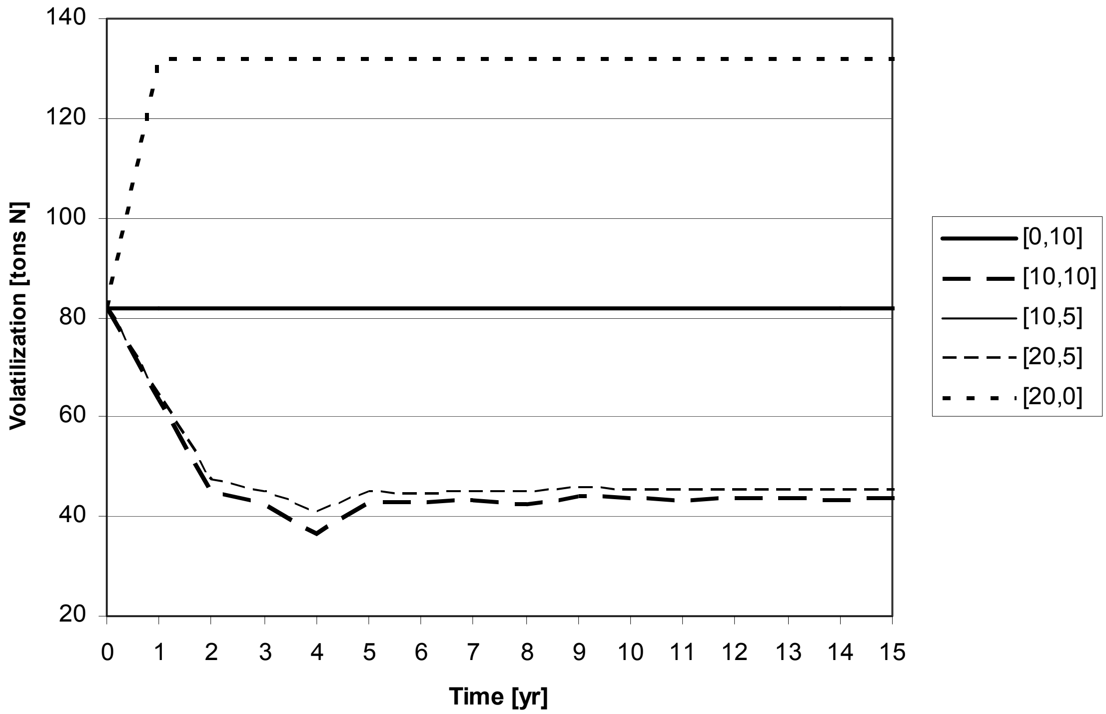

Figure 8: Time path of ammonia volatilization for different emissions tax vectors $\left[w^z, w^v\right]$.

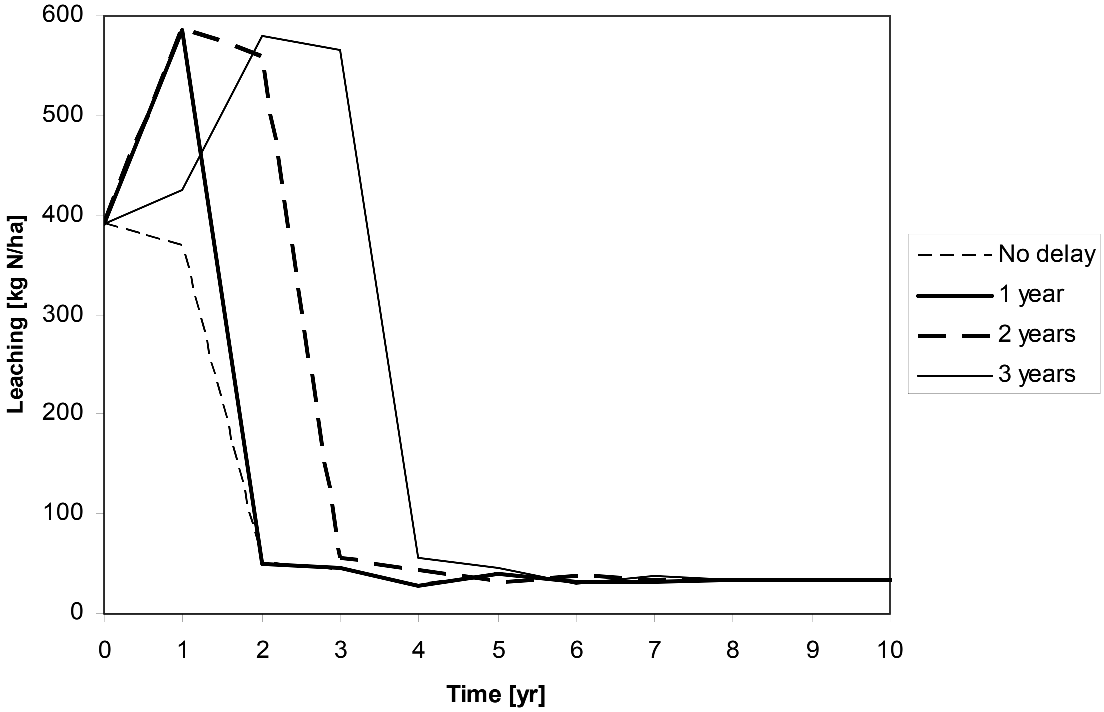

Figure 9: Time path of nitrate leaching for $\left[w^z = 20, w^v = 5 \right]$ and various policy implementation delays.