Effects of Nutrient Restrictions on Confined Animal Facilities: Insights from a Structural Model

Kenneth A. Baerenklau[1]

Nermin Nergis[2]

Kurt A. Schwabe[3]

[1] Assistant Professor
(corresponding author)
Department of
Environmental Sciences
420 Geology Building
University of California
Riverside, CA 92521
V: (951) 827-2628
F: (951) 827-3993
ken.baerenklau@ucr.edu

[2] Formerly: Postdoctoral
Researcher
Department of
Environmental Sciences
University of California
Riverside, CA 92521
nerminnergis@yahoo.com

[3] Associate Professor
Department of
Environmental Sciences
424A Geology Building
University of California
Riverside, CA 92521
V: (951) 827-2361
F: (951) 827-3993
kurt.schwabe@ucr.edu

Selected Paper prepared for presentation at the Western Agricultural Economics Association Annual Meeting, Portland, OR, July 29-August 1, 2007.

Abstract

Nutrient emissions from animal feeding operations continue to degrade water and air quality. New regulations will limit the amounts of nutrients that can be locally applied to land. In this article, a structural-dynamic model of a livestock-crop operation is calibrated with data from a representative farm and is used to predict the effects of nitrogen regulations. Policy simulations clarify the importance of dynamic elements and demonstrate three main results: (1) cost estimates are relatively high; (2) cross-media pollution effects are potentially large; and (3) improved input management appears most promising for reducing both emissions and waste management costs. Implications for policy and future research are discussed.

Key Words: Ammonia, animal feeding operation, cross-media pollution, dairy, dynamic optimization, groundwater, nitrate, nitrogen, nutrient management plan.

Introduction

Over the past 25 years global livestock production has nearly doubled with a trend towards larger and more concentrated operations (FAO 2007). In the United States, the average stocking densities for hog and dairy operations increased from 48 to 912 and from 13 to 115 head per farm, respectively, from 1965 to 2005 (USDA 2006a); similar trends have occurred in the poultry and beef sectors, as well. In lock-step with this trend are increases in the waste by-products from these operations, particularly excess nutrients (Gollehon et al. 2001). Given the potentially negative environmental and health impacts associated with nutrient pollution, animal feeding operations (AFOs) and their by-products have attracted the attention of regulatory agencies and environmental initiatives worldwide (Shortle et al. 2001; Criss and Davidson 2004). Much of this attention has focused on reducing impacts on water quality. For instance, under the Nitrate Directive the European Union requires member states to identify Nitrate Vulnerable Zones (areas where the potential nitrate level in drinking water exceeds 50 mg/l) and produce action plans that target manure waste applications from animal operations (Latacz and Hodge 2003). Such actions are intended to address the fact that the nitrate standard is violated in every EU country, and a large—if not the largest—source of nitrogen leaching is agricultural operations including livestock. In New Zealand, concern over nitrate levels in surface and ground water also has prompted the government to require that dairy farmers implement land-based effluent disposal systems (Cassells and Meister 2001). The Canadian province of Alberta historically has regulated manure application rates based on nitrogen content but is considering implementing stricter phosphorus-based standards (Smith et al. 2006).

In the United States, the largest contributor to lake degradation and third largest contributor to river and stream degradation is nutrient pollution, primarily from agriculture

(Shortle et al. 2001). Nitrate contamination of ground water also is a concern. Nationally, approximately 22% of domestic wells in agricultural regions exceed the federal maximum contaminant level for nitrate (Ward et al. 2005). Previous studies have left little doubt that large AFOs are significant contributors to these problems (Lowry 1987; Mackay and Smith 1990; Harter et al. 2002), especially given the ongoing trend towards consolidation (Shortle et al. 2001; Meyer 2000). In response to these water quality problems, the Clean Water Act was revised in 2003 to regulate large scale animal operators in a manner similar to their European, New Zealand, and Canadian counterparts. Previously, the main focus of the Clean Water Act with regard to large scale confined animal feeding operations (CAFOs) was on restricting the discharge of waste, either directly or through a conveyance system, into water. Under the new amendments, all CAFOs will be required to implement a nutrient management plan (NMP), whereby the application rates for manure must be consistent with agronomic rates of nutrient uptake by crops, and application must be done in a manner that minimizes nitrogen and phosphorus runoff into surface waters (Federal Register 2003).

Unfortunately, as noted by Aillery et al. (2005; p.1), "A logical response by producers operating under a nitrogen-based plan might be to reduce the nitrogen content of manure spread on fields by enabling nitrogen to volatilize into the atmosphere from uncovered lagoons or by applying animal waste to land without incorporating it into the soil." This insight reflects the seminal works of Ayres and Kneese (1969) on residuals management and mass balance and suggests the need to consider coordinated air and water quality policies when addressing this waste management problem. Failure to appreciate the potential response by livestock operators to more stringent water quality regulations and the possible implications of this response for air quality could lead to costly future regulatory adjustments and/or violations of other environmental standards. Such concern is quite legitimate given that animal manure currently is responsible for 33% of all human-related nitrous oxide emissions and 50% of all terrestrial ammonia emissions, both of which contribute to particular matter air pollution and global warming (NRC 2002). And as Ribaudo and Weinberg (2005) note, ammonia emissions in rural areas in the United States already are approaching levels that might trigger Federal action under the Clean Air Act requiring states to regulate these emissions.

The overall goal of this article is to assess the potential effects of nitrogen-based nutrient management plans, both with and without restrictions on ammonia emissions, on large AFOs under a variety of scenarios. We develop a micro-dynamic model of a large dairy operation that is calibrated to a representative farm in the San Joaquin Valley of California. Compared to a baseline simulation that imposes no nutrient restrictions, we evaluate management response to several scenarios including: (i) agronomic restrictions on nitrogen application rates, (ii) agronomic restrictions with improved input management, (iii) agronomic restrictions with improved irrigation efficiency, and (iv) agronomic restrictions with selective culling of animals. We then evaluate the same scenarios with additional restrictions on ammonia emissions.

California's San Joaquin Valley provides a useful test bed for analyzing these issues for several reasons. First, California is the leading US dairy state with nearly 20% of the nation's cows producing approximately 21% of the milk. Its dairy operations are large relative to the national average (763 head per farm vs. 115 head per farm) and therefore indicative of future conditions elsewhere if consolidation continues as expected. Second, between 10 and 15% of California's water supply wells exceed the federal standard for nitrate (Bianchi and Harter 2002), and dairy operations—which can produce nitrate leaching rates ten times greater than from crop-only operations (VanderSchans 2001; Pang et al. 1997)—are significant contributors to the problem (Harter et al. 2002). Third, the San Joaquin Valley Air District, which is home to many of the state's large dairy farms, currently violates federal standards for particular matter air pollution (USEPA 2006). Given that ammonia is a precursor to fine particulate matter air pollution and that approximately 43% of the ammonia emissions in the San Joaquin Valley in 2000 were from dairy farms (Palsgaard 2006), it is apparent that dairy farms contribute to this pollution problem, as well. And fourth, state-level agencies that oversee dairy-intensive regions (e.g., the Central Valley Regional Water Quality Control Board and the San Joaquin Valley Air Pollution Control District) currently are pursuing more effective, albeit uncoordinated, regulations with implications for both producers and the environment.

Related Literature

Research investigating the potential impacts of nutrient management plans on the profitability and waste emissions of AFOs in the U.S. is relatively recent but growing. Ribaudo and Agapoff (2005) estimate that production costs for dairy farms would increase by 0.5-6.5%, and Ribaudo, Cattaneo and Agapoff (2004) similarly estimate that production costs for hog operation would increase by at most 5.5%. Ribaudo et al. (2003) find that implementation costs would range from $0 to $90 per animal unit for dairy operations and from -$5 to $30 per animal unit for hog operations. Huang et al. (2005) report that dairy farms in the southwest with lagoon systems would lose 2 to 4% of net income. For the case of NMP implementation without additional air regulations, Aillery et al. (2005) find that the typical hog operation would lose 5.8% of net returns and that dairy production would decline by less than 1% on average. Kaplan et al. (2004) estimate that livestock and poultry production could decline by as much as 25% in some regions while increasing in others. And Feinerman et al. (2004) derive market welfare losses between 5 and 15%. Collectively these studies present a fairly broad range of possible economic impacts, but much of the variability can be attributed to the type of AFO considered (dairy, swine or poultry), its size and characteristics (e.g., type of manure handling system), the type of NMP (nitrogen or phosphorus), and the amount of off-farm land available for applying manure.

This article extends these results in several dimensions. First, by departing from a static (single period) analysis we are able to address the fact that operator decisions are undertaken in a dynamic framework marked by investment in a capital asset (the herd) and management of a stock (soil nutrients). The dynamic framework imposes on operators additional constraints relative to a static model that may result in higher costs of compliance and/or longer transition periods before pollution reduction goals are achieved. Knowledge of the length of these transition periods and responses at the farm-level to environmental regulations through time can help shape policy maker expectations about alternative waste reduction options. Although some policy guidance gained from static analyses is applicable to dynamic problems, we echo the sentiments expressed in Horan and Shortle (2001) that dynamic analyses can provide additional beneficial insights.[1]

Second, following the literature on irrigated crop production in semi-arid and arid regions (Vickner et al. 1998; Anselin et al. 2004; Schwabe and Knapp 2005), we incorporate a non-uniform irrigation system which allows us to better capture the realities associated with spatially heterogeneous fields and the excess leaching that occurs due to over-application of nutrients. In a recent example that forms the basis for our crop model, 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. Incorporating non-uniform irrigation into our model also allows us to evaluate the impact of improving irrigation efficiency on operator profits and nitrogen emissions.

Third, we incorporate a structural model of lagoon disposal that allows the operator to reduce nitrogen emissions to ground water or surface water by increasing evaporation and volatilization prior to land application (Harris et al. 2001; Massie 2005). Given that evaporation ponds are a common disposal method for saline drainage water in irrigated semi-arid and arid regions throughout the world and that nearly 10% of a sample of San Joaquin Valley dairies utilized evaporative disposal in the mid 1990s (Morse-Meyer et al. 1997), including this option allows for a likely response by large dairies to more stringent regulations that limit nitrogen application rates.

Our specific objectives therefore are threefold: (1) to estimate producer costs with a detailed structural model that captures the dynamic management problem and constraints facing a representative AFO; (2) to revisit the question of pollution reduction, in particular the time required for reductions to be achieved and the potential for cross-media effects; and (3) to advance the modeling techniques used to predict the effects of environmental regulations on AFOs and to evaluate whether the additional model detail and effort produce significantly different results.

A Structural-Dynamic Model of a Dairy Farm

Herd Management

Our model farmer works in discrete time and manages a self-replacing herd of calves, heifers and milk cows.[2] Each year the farmer decides how many animals from each age cohort (a) to retain and how many to sell (cull), and how many replacement heifers to purchase. The equations of motion for the ($\bar{a}$) cohorts can be expressed as a vector function $\mathbf{H}$:

$$\mathbf{h}_{t+1} \equiv \mathbf{H}\left(\mathbf{h}_t, \boldsymbol{\theta}_t, \omega_t, \boldsymbol{\gamma}^h\right), \tag{1}$$

where $\mathbf{h}_t$ is a ($\overline{a}$ x1) vector representing the number of animals in each cohort during year (t); $\boldsymbol{\theta}_t$ is a ($\overline{a}$ x1) vector representing the culling rates; ω_t is the number of replacement heifers purchased; and $\boldsymbol{\gamma}^h$ is a parameter vector describing herd characteristics such as birth and mortality rates.

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. Rotz et al. (1999) list thirty different constituents that may be used by farmers to develop a ration. These constituents exhibit fluctuating availabilities, prices 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, we follow convention and assume each milk cow consumes a fixed cohort-specific ration. Furthermore, because the marginal contributions of each input to milk, meat and waste outputs are largely unknown, we also 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. With this specification, our herd model exhibits constant returns to scale. However, as is common for modern dairies, we also include a herd permit constraint that limits the total number of animal units.

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

$$\pi_t^h \equiv \Pi^h\left(\mathbf{p}^h, \mathbf{x}^h, \mathbf{h}_t, \boldsymbol{\theta}_t, \omega_t, \boldsymbol{\gamma}^h\right), \tag{2}$$

where $\mathbf{p}^h$ is a vector of input and output prices; $\mathbf{x}^h$ is a vector of fixed per-cow inputs and outputs; and the other variables are defined previously.

Waste Management

The second major component of the dairy operation is waste handling and disposal. The amount and composition of waste can vary substantially across farms, depending on the type of housing (e.g., free stall, corral, open lot), manure collection system (e.g., flush, scrape, vacuum), waste treatment (e.g., solids screening, composting, aerobic/anaerobic digestion), waste storage (e.g., lagoons, tanks, stacks), and environmental conditions (e.g., climate). In California's San Joaquin Valley and elsewhere, it is common for large modern dairies to employ free stall housing with waste flushing, solids screening, lagoon storage of liquids, and stacking of dried solids. Solid and liquid wastes are deposited in both the housing structure and the milking parlor and then flushed with water into a solids separator that removes a fraction of the solid content. The separated solids are dried and placed in a manure storage facility; the liquids are stored in an open lagoon. Because this is a typical process for modern dairies and because we have excellent data from a farm like this near Hilmar, California,[3] we specify this type of waste handling system for our model and leave an investigation of alternative systems for future work.

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 of bedding material used; and—because nitrogen is not a conservative pollutant—the residence times in various stages of the waste handling system. Following convention, we assume the farmer cannot affect aspects of the waste handling system that occur between waste generation and storage. Rather, for a given quantity of generated waste (which the farmer affects through herd management decisions), the resulting flows to solid and liquid storage are pre-determined; the farmer then determines how to dispose of the stored waste.

Due to differing transportation costs and marketable end-uses, large dairies often sell dried solid waste but retain liquid waste for irrigating and fertilizing crops. However, NMPs will require farmers to significantly reduce their on-site application rates. The literature cited above suggests that farmers are likely to change their waste management practices by (1) reducing the quantity of stored waste by increasing the ammonia volatilization rate and (2) exporting additional stored waste by paying a custom applicator to haul liquid manure to nearby cropland. We incorporate the first response into our model by allowing the farmer to implement evaporation ponds. We do this for several reasons. First, evaporative disposal already is used by some California dairies similar to our study farm (Morse-Meyer et al. 1997). Second, although nitrogen emissions to ground water and air historically have been treated as separate problems,[4] 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 farmer naturally would attempt to take advantage of the remaining free disposal option before undertaking costly pollution control measures (Aillery et al. 2005; NRC 2002). Third, although there may be other ways to increase ammonia volatilization from a dairy,[5] we note that evaporation of saline drainage water is a well-established, cost-effective waste disposal practice for crop producers in arid and semi-arid regions. Therefore a similar disposal method seems plausible for dairy farmers when faced with stricter nitrate regulations, particularly farmers using the typical waste disposal system we have specified for our model.

We incorporate the second response by specifying an off-site waste disposal cost function that depends on the quantity of exported waste and the distance hauled. Following convention, we assume distance is a function of the suitability and capacity of nearby land for receiving manure nutrients as well as the willingness of the land owners to accept waste. 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 volatilize, be land applied, or be exported off the farm during that season.

Given the preceding, we can incorporate the revenue from dried solid waste, the cost to haul and apply liquid waste, and the cost to install and maintain additional lagoon surface area into a single waste disposal cost function:

$$\pi_t^d \equiv \Pi^d\left(l_{ct}, s_{ct}, e_t, \mathbf{p}^d, \boldsymbol{\gamma}^d\right), \tag{3}$$

where l_{ct} and s_{ct} are the amounts of liquid and solid wastes applied at the dairy; e_t is the total surface area of the lagoons; $\mathbf{p}^d$ is a vector of unit costs; and $\boldsymbol{\gamma}^d$ is a parameter vector including information about the characteristics of the stored waste and the receiving land.

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. A notable aspect of this model component is the uniformity of the irrigation system which has been shown to significantly affect soil nitrogen levels and nitrate leaching rates (Schwabe and Knapp 2005) but which has been absent from previous studies of livestock-crop operations. Irrigation system uniformity is captured by a parameter $\beta \in [0,\infty]$ which represents the water infiltration coefficient (i.e., the fraction of applied water that infiltrates into the root zone) at each point in the field and which has distribution $g(\beta)$ per unit area. We can therefore specify the equations of motion for the soil nitrogen concentrations at any point in the field as a vector function $\mathbf{N}$:

$$\mathbf{n}_{ct+1}(\beta) \equiv \mathbf{N}\left(\mathbf{n}_{ct}(\beta), s_{ct}, l_{ct}, f_{ct}, i_{ct}, \boldsymbol{\gamma}^n\right), \tag{4}$$

where $\mathbf{n}_{ct}\left(\beta\right)$ is a (2x1) vector of organic and inorganic soil nitrogen concentrations; s_{ct}, l_{ct}, f_{ct}, and i_{ct} are control variables representing the amounts of solid waste, liquid waste, commercial fertilizer and irrigation water applied to fields; and $\boldsymbol{\gamma}^{n}$ is a parameter vector.[6] Applications of liquid waste also are subject to a constraint that they must be sufficiently diluted with irrigation water in order to avoid damaging crops with high concentrations of waste components that do not volatilize (e.g., salts) and therefore become concentrated in the residual lagoon water (Swenson 2004).

Crop production at any point in the field can be expressed similarly as a function Y:

$$y_{ct}\left(\beta\right) \equiv Y\left(\mathbf{n}_{ct}\left(\beta\right), s_{ct}, l_{ct}, f_{ct}, i_{ct}, \boldsymbol{\gamma}^{y}\right), \tag{5}$$

where $\boldsymbol{\gamma}^{y}$ is a parameter vector. Nitrogen leaching and ammonia volatilization from any point in the field also can be expressed as functions of the same state and control variables. Aggregate crop yields are calculated by integrating Y over $g\left(\beta\right)$ and multiplying by the total cropped area; aggregate amounts of leaching and volatilization are calculated similarly.

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

$$\pi_{ct}^{y} \equiv \Pi^{y}\left(\mathbf{p}^{y}, \mathbf{x}^{y}, \mathbf{n}_{ct}\left(\beta\right), s_{ct}, l_{ct}, f_{ct}, i_{ct}, e_{t}, \boldsymbol{\gamma}^{n}, \boldsymbol{\gamma}^{y}\right), \tag{6}$$

where $\mathbf{p}^{y}$ is a vector of input and output prices; $\mathbf{x}^{y}$ is a vector of fixed inputs to the cropping system; and the other variables have been defined previously.

Optimization

Defining $\pi_{t} \equiv \pi_{t}^{h} + \sum_{c} \pi_{ct}^{y} - \pi_{t}^{d}$, collecting all prices into a vector $\mathbf{p}$ and all parameters (including fixed inputs and outputs) 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:

$$\max_{\{\boldsymbol{\theta}_t, s_{ct}, l_{ct}, f_{ct}, i_{ct}, \omega_t, e_t\}} \left[\sum_{t=0}^{T} \rho^t \pi_t \left(\mathbf{h}_t, \mathbf{n}_{ct}(\beta), \boldsymbol{\theta}_t, s_{ct}, l_{ct}, f_{ct}, i_{ct}, \omega_t, e_t \mid \mathbf{p}, \boldsymbol{\Gamma} \right) \right], \tag{7}$$

subject to the equations of motion for the herd and the soil nitrogen concentrations, constraints on total available land and total allowable animal units, mass balance constraints on solid and liquid waste streams, and the liquid waste dilution constraint. This statement defines an optimal control problem with state variables for the herd age cohorts and soil nitrogen concentrations, and with control variables for the culling rates, the application rates for solid waste, liquid waste, chemical fertilizer, and irrigation water, the number of purchased replacement heifers, and the evaporation pond area. We solve this dynamic optimization problem in GAMS as a constrained non-linear programming problem (Standiford and Howitt 1992) utilizing the CONOPT solver.

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 and sensitivity analyses. To find feasible starting values for the 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 large enough to avoid boundary effects). We then solve the dynamic problem using the period-by-period solution as the starting values, check if the model 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 1 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, the model appears to be calibrated well. Animal cohort numbers are similar to those reported by VanderSchans (2001) for the Hilmar site. Differences are most likely due to off-farm rearing of some 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 per cow is low compared to their estimate, 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 $1,239/cow, which is very close to their estimate. However, we retain the lower values because they are much closer to the reported average for California dairies (USDA 2006b).

Ammonia volatilization from our model farm is similar to reported values, and nitrate leaching is nearly identical to VanderSchans' best estimate (based on a hydrologic model) for the Hilmar farm. Corn and wheat yields are high but within reason, as are the concentrations of nitrogen in the manure storage lagoon (all of which are compared to other published sources due to lack of data for the Hilmar farm). Applied water (irrigation plus lagoon water) is close to the Hilmar farm estimate, but applied chemical (nitrogen) fertilizer is significantly different. Our model farm does not apply any chemical fertilizer, which supports results 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. However, the only noteworthy changes derived from imposing the midpoint application rate of 205 kg N/ha-yr on our model is a 1% decrease in profit and a 7% increase in the leaching rate. Lastly, and consistent with VanderSchans (2001), our model farm sells and exports all dried solid manure.

Nutrient Management Plan Simulations

Nutrient management plans are readily incorporated into our modeling framework by specifying an additional constraint that limits the amount of nitrogen that may be land applied each year at the dairy. Following convention, the land application constraint is set equal to the estimated total amount of nitrogen contained in the harvested portions of the cropping system, plus an allowance for unavoidable soil nitrogen losses. To make our constraint consistent with previous studies, quantities of harvested nitrogen are based on crop-specific nutrient uptake rates published by Lander et al. (1998), and the allowance for unavoidable losses is taken from Kellogg et al. (2000). This gives a maximum nitrogen application rate of 412 kg N/ha-yr, whereas the total amount of applied nitrogen in the unregulated steady state is 2196 kg N/ha-yr.

Our policy simulations assume the dairy farm is initially at the steady state operating position derived in the model calibration section. We then introduce the NMP constraint and we derive the dynamically optimal response for the dairy. We focus on the change in the net present value (NPV) of farm operations during the simulated time period, as well as the time paths for three variables: herd size [number of milk cows], nitrate leaching [kg N/ha-yr], and ammonia volatilization [kg N/yr]. Again following convention, we present the results for different levels of "willingness to accept manure" (WTAM) by surrounding land operators. WTAM is the percentage of surrounding land suitable for receiving manure that is willing to accept it. For our study site, we calculate that 25% of surrounding land is suitable for receiving manure (Kellogg et al. 2000, USDA 2006c); the WTAM values we consider therefore correspond to 25%, 15%, 5% and 1% of surrounding land.[7]

Scenario 1 in table 2 shows the policy-induced NPV loss and new steady state levels for the other variables of concern given our baseline model parameter values. The predicted loss ranges from 12 to 19% of NPV, depending on WTAM. Previous estimates for implementing nitrogen-based NMPs at "large" dairy operations (typically ≥ 700 cows) are in the range of 2-6% of profits (Ribaudo et al. 2003, Ribaudo and Agapoff 2005, Huang et al. 2005, Aillery et al. 2005). Whereas these studies focus on off-site manure disposal, our estimate includes 2.3% from reduced production (lower crop yields due to less applied water and nitrogen), 4.7% from efforts to increase ammonia volatilization, and 5-12% from additional off-site waste disposal. Although this result confirms that off-site disposal of manure will be a key response to NMP requirements, it does not support the notion that a simpler analysis focusing on waste disposal costs alone will be sufficient for estimating the economic implications for producers. We revisit this finding and discuss additional implications in the concluding section.

The other variables in table 2, which characterize the new steady state operating position of the dairy, are not affected by WTAM in this scenario. Relative to the unregulated steady state, the herd size remains unchanged at 1,445 milk cows, the leaching rate falls from 413 to 6 kg N/ha-yr, and the volatilization rate increases from 82,463 to 130,569 kg N/yr. Figure 1 shows that the leaching rate falls precipitously during the first year and then much more gradually thereafter (note the logarithmic scale). After 4 years the leaching rate is still twice as high as the eventual steady state value, but after 8 years it is within 10% of this value. These results are consistent with the literature on nitrate leaching from crop operations (Schwabe and Knapp 2005) and, together with the result for the herd size, suggest that the dynamics of NMP implementation in this scenario are primarily captured by the crop production component of the model rather than the herd component. However, we will see that culling decisions play a more prominent role when NMPs are implemented in conjunction with ammonia regulations.

Finally, we observe a 58% increase in volatilization of ammonia emissions for this scenario. The increase in ammonia emissions is substantially larger than the only comparable estimate we can find elsewhere (for hog operations, by Aillery et al. 2005), and is likely due to the additional control variable in our model which allows the farmer to increase lagoon surface area. Apparently this is a low-cost response to NMP requirements that can produce a significant increase in ammonia emissions; in fact, our model predicts that farmers will maximize lagoon emissions for all values of WTAM. Figure 1 shows that the time path of ammonia emissions is qualitatively similar to that for nitrate leaching: the new steady state value is attained during the first year of NMP implementation with no additional increases thereafter.

Sensitivity Analysis

Similar to previous studies, the preceding analysis does not account for the possibility that, when faced with new waste disposal restrictions, farmers may attempt to implement (currently unproven) input management practices in an effort to reduce costs. For example, research suggests that the nitrogen concentration of the waste stream may be reduced 20-40% by feeding amino acid supplements (Kohn 1999), 8-15% by grouping and feeding cows according to milk production levels (Castillo 2003), and nearly 10% by adjusting the composition of the feed ration (Jonker et al. 2002). Dunlap et al. (2000) estimate that feeding bovine growth hormone, milking three times daily, and exposing cows to artificial daylight during nighttime collectively can reduce waste nitrogen by 16%. To the extent these practices are currently used by California dairies, our model implicitly accounts for their impacts on milk production and waste generation because we calibrate our model with state-wide averages. Assuming none is widely used, the nutrient content of the waste stream could be approximately halved if all of these practices were implemented. However, a significant (and still largely unknown) cost would be incurred either by the farmer or by an agency offering adoption subsidies for these practices. To conduct a sensitivity analysis, we assume our model farm adopts all of these fully-subsidized practices (i.e., at no cost) and achieves a 50% reduction in the nitrogen concentration of the waste stream.

Scenario 2 of table 2 presents these policy simulation results. Relative to scenario 1, adopting these practices saves the farmer 2-6% of net income depending on WTAM. Whether or not these gains would offset adoption costs in the absence of government subsidies is a question we currently cannot answer; here we consider the effect on steady state nitrogen emissions. Relative to the baseline policy simulations, halving the nitrogen concentration of the waste stream reduces ammonia emissions by 49% but *increases* nitrate leaching from 6.0 to 8.6 kg N/ha-yr.[8] The increased leaching arises from multiple effects. First, with a lower nitrogen concentration in the waste, more waste is retained on the farm for land application. Second, because this waste contains the same concentration of salts as it did in the baseline case, relatively more irrigation water (about 10%) must be applied to achieve sufficient dilution. This additional water flushes more nitrates through the soil and increases the leaching rate.

This somewhat surprising result suggests that the problem of nitrogen emissions should not be considered as a simple nutrient mass-balance problem, but rather as a more complicated problem involving relationships between nutrients, water and other waste components.[9] It also suggests that improved irrigation uniformity could allow the NMP constraint to be relaxed without increasing the leaching rate because less water would pass through the rootzone and into the aquifer. In fact, assuming perfectly uniform irrigation, our model predicts that the NMP constraint could be increased from 412 to 1,200 kg N/ha-yr while still achieving 6 kg N/ha-yr of nitrate leaching, yet at the expense of higher ammonia emissions from crop fields. The associated NPV loss would be reduced to 6-8% of net income, depending on WTAM, without any improvements to input management. These results are summarized as the third scenario in table 2; policy implications are discussed later.

Another management alternative overlooked by the existing literature (and our baseline scenario) is that of selectively culling lower producing animals when faced with waste disposal restrictions, which also would tend to reduce NMP implementation costs relative to the case of homogenous age cohorts. Although culling models do exist (e.g., Van Arendonk 1985), they have not been used in the context of environmental pollution control. We use our model to approximate such culling decisions by introducing cohort-specific milk yield distributions and assuming farmers cull the lowest yielding cows first. Specifically, we assume each cohort milk yield distribution is uniform with mean given by the cohort-specific milk yield used in the baseline scenario and with the highest yielding cow producing twice as much as the lowest yielding cow.[10] This gives a slightly different unregulated steady state operating position for the farm: profits are 13% higher, the herd contains 1,392 milk cows, leaching is 404 kg N/ha-yr, and volatilization is 82,358 kg N/yr. Scenario 4 of table 2 presents the policy simulation results relative to these unregulated steady state values. The response of the dairy for all WTAM values is similar to the response in scenario 1 which assumed a homogenous herd: the herd size remains unchanged, leaching drops substantially, and volatilization increases by 58%. Interestingly, the ability to cull low yielding cows reduces the percentage income loss by only 2-3% relative to scenario 1, suggesting that such decisions may not play a major role in NMP implementation.

NMP Simulations with Air Regulations

Given our predictions of substantial policy-induced increases in ammonia volatilization and the documented air quality problems in livestock-intensive regions, we now consider the likely effects of implementing ammonia regulations in addition to NMPs. Regulations on ammonia emissions could take a variety of forms; as in Aillery et al. (2005), we consider the relatively straight-forward case of a quantity restriction. The regulation we consider requires that total ammonia emissions from the farm not exceed the unregulated steady-state level. This may be a relatively lenient restriction, given that air quality regulators in California are pursuing strategies to *reduce* ammonia emissions from AFOs.

Policy simulation results for the same scenarios considered above are given in table 3. The second scenario (improved input management) is identical to that of table 2 because the optimal strategy for this scenario without air regulations is to reduce volatilization below the unregulated steady state value; therefore the additional air quality regulation is not binding. However, the results for the other scenarios are significantly different from those in table 2. For the baseline parameter values the expected loss is now much higher at 37-45% of net farm income, depending on WTAM. These estimated losses are about 2-3 times as high as comparable estimates in the existing literature (Aillery et al. 2005). With restrictions on both waste streams, table 3 shows it is now optimal to reduce the herd size and incur both crop and livestock production losses in scenarios 1, 3 and 4. Though not shown graphically, herd reductions are qualitatively similar to nitrate leaching reductions: large reductions occur during the first 1-2 years, followed by smaller reductions (and sometimes small cyclical fluctuations) thereafter. In scenarios 1 and 4 the associated production losses represent a large portion of the total loss: 15-35% of net farm income depending on WTAM. In scenario 3 these production losses range from 3-21% of the total. Selective culling again does not have a large effect on costs, and improved irrigation uniformity has a relatively smaller effect than it does in the absence of air regulations.

Discussion and Conclusions

Economies of scale and technological innovation are resulting in more concentrated animal feeding operations worldwide. Governments are reacting to the associated waste management problem primarily with tighter restrictions on nutrient application rates to protect water quality. However, a potentially perverse outcome from these more stringent nutrient restrictions is an incentive to increase volatilization of nitrogen, often in areas located near population centers and/or in areas where air quality already is degraded (FAO 2007).

The present study focuses on the dairy industry, which increasingly has been the target of nutrient management plans in the European Union, New Zealand, Canada, and the U.S. We develop a structural-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 the San Joaquin Valley and with additional data from other sources. The optimized characteristics of the farm, including herd size, crop yields, amounts of applied water, nitrate leaching, ammonia volatilization, and net farm income are consistent with available comparison data.

Regarding our first objective – to estimate producer costs with a detailed structural model that captures the dynamic management problem and constraints facing a representative AFO – we find that implementing nitrogen-based NMPs could generate profit losses around 12-19%, substantially greater than the most comparable estimates from previous studies. While more work is needed to clarify the exact sources of the differences (e.g., production scales and technologies, modeling frameworks), it is apparent that NMP implementation costs for large dairies could be relatively high. Expected losses around 12-19% could make policy implementation difficult in some regions or induce unanticipated changes in the industry (e.g., restructuring, relocation). Indeed, given that the impacted producers operate relatively larger farms and produce a large share of total output, their operating decisions could have non-trivial effects on local economics, markets, and even trade (Cassells and Meister 2001). Overall we think NMP implementation will have a greater economic impact on large producers than has been suggested by previous farm-level studies.

In terms of how these costs might be reduced, our simulations suggest two promising avenues: improved input management and irrigation uniformity. According to our estimates in table 3, improved input management has the potential to reduce economic losses by 75% when both NMPs and air regulations are implemented together. However, this finding is based on assumptions about currently unproven technologies and the costs producers might incur to adopt them. It also comes with the caveat that nitrate leaching may actually increase as the nitrogen throughput of an AFO decreases; but this observation simply reinforces our belief that regulating the application of nitrogen alone is not the best approach to the problem. Regardless, research is needed to develop these technologies, identify the associated cost functions, and examine what types of additional incentives – if any – might be appropriate for encouraging their use.

Improved irrigation uniformity also could reduce implementation costs *if NMP restrictions are relaxed accordingly*; however, to our knowledge such allowances currently are not being considered. By regulating nitrogen application rates rather than leaching rates, regulators are missing an opportunity to encourage producers to adopt less polluting and potentially cost-saving irrigation systems. This is a case of regulating a precursor to pollution rather than the pollution itself, which typically produces an inefficient outcome. An incentive could be created, for example, if the NMP constraint were related to the irrigation system choice such that users of more uniform systems were allowed to apply more nitrogen.

Regarding our second objective – to revisit the question of pollution reduction, in particular the time required for reductions to be achieved and the potential for cross-media effects – we find that initial reductions in nitrate leaching will occur quickly but achieving steady state levels will require 7-9 years. We also predict that ammonia emissions will increase rapidly and there is considerable risk of substantially degrading air quality if NMPs are implemented without ammonia regulations. These results differ from recent work by Aillery et al. (2005), who find a notably smaller potential for cross-media pollution from hog operations, and suggest more research is needed to assess the trade-off and determine what might be done to manage it. Issues to consider include the benefits obtained from reducing emissions, including the temporal aspect of exposure to both nitrate and ammonia: whereas ammonia emissions can have an immediate effect on air quality, nitrate emissions may take longer to migrate through the hydrologic system before impacting a recreational resource or a drinking water source. Such an analysis also should consider that ammonia alone does not create airborne particulate matter but rather must interact with sulfur or nitrogen oxides which primarily are the result of combustion processes. Given the high cost we estimate to implement both water and air regulations, increased ammonia emissions may be deemed acceptable in regions that are oxide-limited. Population and climate variables also will affect this tradeoff, and it is likely that populous arid regions that rely on ground water resources will face the most difficult choices.

Lastly, regarding our third objective – to advance the modeling techniques used to predict the effects of environmental regulations on AFOs and to evaluate whether the additional model detail and effort produce significantly different results – we find somewhat mixed results. On the one hand, the differences between our results and those of previous studies, as well as the additional temporal insights generated by a multi-period framework, suggest that structural dynamic modeling of AFO regulation should not be dismissed as "not worth the trouble." More work is needed to clarify the exact sources of the differences and to determine if other potentially important aspects of the problem (i.e., the waste dilution constraint, irrigation system uniformity) have been overlooked. A formal comparative modeling analysis is beyond the scope of this work because existing static models cannot readily be characterized as constrained versions of our dynamic model, but such an analysis would be a useful next step. On the other hand, we also find that herd management dynamics are not as important as soil nitrogen dynamics in much of the present analysis. Most likely this is because each age cohort can be controlled (culled) separately, which effectively relaxes the constraints imposed by the state equations and makes the herd management component behave more like a static optimization problem. A simpler approach that still includes soil nitrogen dynamics but omits the formal state equations for the herd age cohorts while still allowing the operator to choose a herd size might be an appropriate compromise between fully static and dynamic models.

Endnotes

[1] Previous studies have incorporated dynamic elements when examining livestock management decisions (e.g., Tozer and Huffaker 1999; Chavas and Klemme 1986; Van Arendonk 1985) but not in the context of environmental regulation. The only dynamic analysis of livestock production and environmental regulation that we are aware of is Schnitkey and Miranda (1993). 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.

[2] Due to the level of detail, much of the model exposition is contained in a referee's appendix available from the authors upon request. This includes parameter values and functional forms for the Herd Management, Waste Management, and Crop Management model components. In the main text we present the important variables and relationships that are necessary for understanding our general approach.

[3] VanderSchans (2001) provides a detailed description of the study farm.

[4] 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).

[5] For example, lagoon aeration; but this approach does not appear to be very effective (Rumburg et al. 2004, Zhao and Chen 2003).

[6] Here we use the subscript $ct+1$ as shorthand notation for the next cropping season, which could be either the next season of the same year or the first season of the next year.

[7] Our study site is located in an area where off-site disposal of manure should be relatively cheap. A relatively large share of the surrounding land is intensively farmed and able to

receive substantial quantities of waste nitrogen (Kellogg et al. 2000). Therefore the NMP implementation cost for our model farm will tend to be less than for a similar farm facing competition for land from other AFOs, high-value agricultural producers, or urban developers. Furthermore, because we use straight-line distances to calculate hauling costs, our disposal cost estimates will tend to be less than those for an actual dairy.

[8] Kaplan and Johansson (2003) derive a similar result using a different modeling approach.

[9] The observation that water application rates are an important component of the nitrate leaching problem is consistent with the findings of Schwabe and Knapp (2005).

[10] Available data on within-herd milk yield variability is limited. Cassel (2001) reports that one rating system classifies cows into five groups, with the highest producing at least 110% of the herd average and the lowest ("probable cull cows") producing less than 80% of the average. Several sources (e.g., Wattiaux 2003) suggest the distribution is approximately normal. Our assumptions therefore are optimistic: the variability is somewhat larger than in Cassel (2001) and there are relatively more cows in the tails of the distribution which translates into larger potential efficiency gains from culling.

References

Abernathy, K., Bailey, D., Borba, G., Byers, M., Cativiela, J., deBoom, N., Descary, B., Gill, S., Hamilton, K., Martin, P., Meyer, D., Mullinax, D., Shears, J., Simunovic, C., Sweet, J., Warner, D., Watson, J. 2006. Recommendations to the San Joaquin Valley Air Pollution Control Officer Regarding Best Available Control Technology for Dairies in the San Joaquin Valley. Available from URL: http://www.valleyair.org/busind/pto/dpag/dpag_idx.htm [accessed 2005].

Adriano, D., Chang, A., and Sharpless, R. 1974. Nitrogen Loss from Manure as Influenced by Moisture and Temperature, *Journal of Environmental Quality* 3(3), 258-261.

Aillery, M., Gollehon, N., Johansson, R., Kaplan, J., Key, N., and Ribaudo, M. 2005. Managing Manure to Improve Air and Water Quality, United States Department of Agriculture, Economic Research Service Economic Research Report No. 9.

Anselin, L., Bongiovanni, R., Lowenberg-DeBoer, J. 2004. A Spatial Econometric Approach to the Economics of Site-Specific Nitrogen Management in Corn Production, *American Journal of Agricultural Economics* 86, 675-687.

Ayres, R. and Kneese, A. 1969. Production, consumption and externalities, *American Economic Review* 59, 282-97.

Bianchi, M. and Harter, T. 2002. Nonpoint Sources of Pollution in Irrigated Agriculture. Publication 8055. University of California, Division of Agriculture and Natural Resources. Available from URL: http://groundwater.ucdavis.edu/Publications/Harter_FWQFS_8055.pdf [accessed 2006].

Brittan, K., Munier, D., Schmierer, J., Klonsky, K., and Livingston, P. 2004. 2004 Wheat Cost and Return Study – Sacramento Valley, University of California Cooperative Extension

Publication WH-SV-04. Available from URL: http://coststudies.ucdavis.edu/ [accessed 2005].

Campbell Mathews, M. 2006. Impact of Improved Nutrient Management Practices on Shallow Groundwater Quality under Dairies, *Nitrate in California's Groundwater: Are We Making Progress?* Paper presented at the Groundwater Resources Association Symposium, 4-5 Apr 2006, Modesto, CA.

Cassel, B. 2001. Using DHI records to make culling decisions: Lactation Ratings, ERPA's, and Predicted Producing Abilities, Blacksburg, VA: Virginia Cooperative Extension Publication Number 404-083. Available from URL: http://www.ext.vt.edu/pubs/dairy/404-083/404-083.html [accessed 2005].

Cassells, S., and Meister, A. 2001. Cost and trade impacts of environmental regulations: effluent control and the New Zealand dairy sector, *The Australian Journal of Agricultural and Resource Economics* 45(2), 257-274.

Castillo, A. 2003. Nitrogen Excretion in Dairy Cows: Effects of Grouping Animals and Balancing Diets, University of California Cooperative Extension Service Newsletter. Available from URL: http://cemerced.ucdavis.edu/newsletterfiles/Dairy_Notes3170.pdf [accessed 2005].

Chang, A.C., T. Harter, J. Letey, D. Meyer, R., Campbell Mathews, M., Mitloehner, F., Pettygrove, S., Robinson, P., and Zhang, R. 2004. Managing Dairy Manure in the Central Valley of California. Draft Report by the University of California Committee of Consultants on Dairy Waste Management. Department of Environmental Sciences, University of California – Riverside.

Chang, A.C., T. Harter, J. Letey, D. Meyer, R., Campbell Mathews, M., Mitloehner, F., Pettygrove, S., Robinson, P., and Zhang, R. 2005. Managing Dairy Manure in the Central Valley of California, Report by the University of California Committee of Experts on Dairy Manure Management. URL: http://www.waterboards.ca.gov/centralvalley/available_documents/index.html#confined [accessed 2006].

Chavas, J., and Klemme, R. 1986. Aggregate Milk Supply Response and Investment Behavior on U.S. Dairy Farms, *American Journal of Agricultural Economics* 68, 55-66.

Criss, R., and Davidson, M. 2004. Fertilizers, Water Quality, and Human Health, *Environmental Health Perspectives* 112(10), A536.

Crohn, D. 1996. Planning Biosolids Land Application Rates for Agricultural Systems, *Journal of Environmental Engineering* 122, 1058-1066.

Dunlap, T., Kohn, R., Dahl, G., Varner, M., and Erdman, R. 2000. The Impact of Somatotropin, Milking Frequency, and Photoperiod on Dairy Farm Nutrient Flows, *Journal of Dairy Science* 83, 968-976.

Smith, E., Card, G., and Young, D. 2006. Effects of Market and Regulatory Changes on Livestock Manure Management in Southern Alberta, *Canadian Journal of Agricultural Economics* 54(2), 199- 213.

Food and Agricultural Organization (FAO). 2007. Pollution from Industrialized Livestock Production. Livestock Policy Brief 2, Rome, Italy. Available from URL: http://www.fao.org/ag/aga.html [accessed 2007].

Federal Register. 2003. National Pollutant Discharge Elimination System Permit Regulation and Effluent Limitation Guidelines and Standards for Concentrated Animal Feeding Operations (CAFOs); Final Rule, 68(29), 7175-7184.

Feinerman, E., Bosch, D., and Pease, J. 2004. Manure Applications and Nutrient Standards, *American Journal of Agricultural Economics* 86, 14-25.

Gollehon, N., Caswell, M., Ribaudo, M., Kellogg, R., Lander, C., and Letson, D. 2001. Confined Animal Production and Manure Nutrients, *Agriculture Information Bulletin AIB771*, United States Department of Agriculture, Economic Research Service.

Hacker, R. and Du, Z. 1993. Nitrogen Flow in Pig Production and Environmental Consequences, *EAAP Publication No. 69*, 8-11 June 1993, Wageningen, The Netherlands.

Harris, B., Hoffman, D., and Mazac Jr, F. 2001. Reducing Contamination by Improving Milking Center Wastewater Treatment. Texas A&M University Blackland Research Center. Available from URL: http://waterhome.brc.tamus.edu/texasyst/t6032a.html [accessed 2005].

Harter, T., Mathews, M., and Meyer, R. 2001. Effects of Dairy Manure Nutrient Management on Shallow Groundwater Nitrate: A Case Study, Paper No. 01-2192. Presented at the 2001 American Society of Agricultural Engineers Annual Meetings, Sacramento, CA.

Harter, T., Meyer, R., and Mathews, M. 2002. Nonpoint source pollution from animal farming in semi-arid regions: Spatio-temporal variability and groundwater monitoring strategies, in Ribeiro, L. (ed.), *Future Groundwater Resources at Risk*, Proceedings of the 3rd International Conference, June 2001, Lisbon, Portugal, pp. 363-372.

Horan, R. and Shortle, J. 2001. Environmental Instruments for Agriculture, in Shortle, J. and Abler, D. (eds), *Environmental Policies for Agricultural Pollution Control*, CAB International. New York.

Huang, W., Magleby, R., and Christensen, L. 2005. Economic Impacts of EPA's Manure Application Regulations on Dairy Farms with Lagoon Liquid Systems in the Southwest Region. *Journal of Agricultural and Applied Economics* 37, 209-227.

Jonker, J., Kohn, R., and High, J. 2002. Dairy Herd Management Practices that Impact Nitrogen Utilization Efficiency, *Journal of Dairy Science* 85, 1218-1226.

Kaplan, J., Johansson, R., and Peters, M. 2004. The Manure Hits the Land: Economic and Environmental Implications When Land Application of Nutrients is Constrained, *American Journal of Agricultural Economics* 86, 688-700.

Kellogg, R., Lander, C., Moffitt, D., and Gollehon, N. 2000. Manure Nutrients Relative to the Capacity of Cropland and Pastureland to Assimilate Nutrients: Spatial and Temporal Trends for the U.S. USDA NRCS Report No. NPS00-0579. Washington, DC.

Kim, C., Hostetler, J., and Amacher, G. 1993. Regulation of Groundwater Quality with Delayed Responses, *Water Resources Research* 29, 1369-1377.

Kohn, R. 1999. The Impact of Dairy Herd Management on Nutrient Losses to Water Resources, University of Maryland Cooperative Extension Service. Available from URL: http://www.agnr.umd.edu/nutrients/ [accessed 2005].

Lander, C., Moffitt, D., Alt, K. 1998. Nutrients Available from Livestock Manure Relative to Crop Growth Requirements, USDA NRCS Resource Assessment and Strategic Planning Working Paper 98-1. Washington, D.C.

Latacz-Lohmann, U. and Hodge, I. 2003. European agri-environmental policy for the 21st century. *Australia Journal of Agricultural and Resource Economics* 47(1), 123-139.

Lowry, P. 1987. Hilmar Ground Water Study, Regional Water Quality Control Board Central Valley Region, Sacramento, California.

Mackay, D., and Smith, L.A. 1990. Agricultural chemicals in groundwater: Monitoring and management in California, *Journal of Soil and Water Conservation* 45, 253-255.

Massie, L. 2005. Reducing the Risk of Groundwater Contamination by Improving Milking Center Wastewater Treatment, Farmstead Assessment System Fact Sheet #10, New Mexico State University. Available from URL: http://www.cahe.nmsu.edu/farmasyst/pdfs/10fact.pdf [accessed 2006].

Meyer, D. 2000. Dairying and the Environment. *Journal of Dairy Science* 83, 1419-1427.

National Research Council. 2002. The Scientific Basis for Estimating Air Emissions from Animal Feeding Operations. Washington, DC: National Academy Press.

Nkonya, E., and Featherstone, A. 2000. Determining Socially Optimal Nitrogen Application Rates Using a Delayed Response Model: The Case of Irrigated Corn in Western Kansas, *Journal of Agricultural and Resource Economics* 25, 453-467.

Pang, X., Letey, J., and Wu, L. 1997. Irrigation Quantity and Uniformity and Nitrogen Application Effects on Crop Yield and Nitrogen Leaching, *Soil Science Society of America Journal* 61, 257-261.

Palsgaard, J. 2006. Merced County Animal Confinement Ordinance Revisions and EIR. Available from URL: http://www.eco-farm.org/sa/sa_dairy_syn_merced_co_CAO.html [accessed 2006].

Ribaudo, M. and Agapoff, J. 2005. Importance of Cost Offsets for Dairy Farms Meeting a Nutrient Application Standard, *Agricultural and Resource Economics Review* 34, 173-184.

Ribaudo, M., Cattaneo, A., and Agapoff, J. 2004. Cost of Meeting Manure Nutrient Application Standards in Hog Production: The Roles of EQIP and Fertilizer Offsets, *Review of Agricultural Economics* 26, 430-444.

Ribaudo, M., Gollehon, N., Aillery, M., Kaplan, J., Johansson, R., Agapoff, J., Christensen, L., Breneman, V., and Peters, M. 2003. Manure Management for Water Quality: Costs to

Animal Feeding Operations of Applying Manure Nutrients to Land. Agricultural Economic Report AER824. United States Department of Agriculture, Economic Research Service.

Ribaudo, M. and Weinberg, M. 2005. Improving Air and Water Quality can be Two Sides of the Same Coin, *Amber Waves* September, 35-41.

Rotz, C., Mertens, D., Buckmaster, D., Allen, M., and Harrison, J. 1999. A Dairy Herd Model for Use in Whole Farm Simulations, *Journal of Dairy Science* 82, 2826-2840.

Rotz, C., Oenema, J., and van Keulen, H. 2003. Whole-Farm Management to Reduce Nitrogen Losses from Dairy Farms, Paper no. 032154 presented at the 2003 ASAE International Meeting, July 27-30, 2003. Las Vegas, Nevada.

Rumburg, B., Neger, M., Mount, G., Yonge, D., Filipy, J., Swain, J., Kincaid, R., and Johnson, K. 2004. Liquid and Atmospheric Ammonia Concentrations from a Dairy Lagoon during an Aeration Experiment, *Atmospheric Environment* 38, 1523-1533.

San Joaquin Valley Air Pollution Control District (SJVAPCD). 2006. Ambient Air Quality Standards & Valley Attainment Status. Available from URL: http://www.valleyair.org/aqinfo/attainment.htm [accessed 2006].

Schnitkey, G., and Miranda, M. 1993. The Impacts of Pollution Controls on Livestock-Crop Producers, *Journal of Agricultural and Resources Economics* 18, 25-36.

Schwabe, K., and Knapp, K. 2005. Nitrogen as a Capital Input and Stock Pollutant: A Dynamic Analysis of Corn Production and Nitrogen Leaching under Non-Uniform Irrigation, Paper presented at the American Agricultural Economics Association 2005 Annual Meetings, July 24-27, 2005, Providence, RI.

Shortle, J., Abler, D., and Ribaudo, M. 2001. Agriculture and Water Quality: The Issues, in Shortle, J. and Abler, D. (eds), Environmental Policies for Agricultural Pollution Control, CAB International. New York.

Standiford, R., and Howitt, R. 1992. Solving Empirical Bioeconomic Models: A Rangeland Management Application, *American Journal of Agricultural Economics* 74, 421-433.

Tozer, P., and Huffaker, R. 2000. Dairy Deregulation and Low-Input Dairy Production: A Bioeconomic Evaluation, *Journal of Agricultural and Resource Economics* 24, 155-172.

United States Department of Agriculture (USDA). 2006a. Quick Stats agricultural statistics database. National Agricultural Statistics Service. Available from URL: http://www.nass.usda.gov/ [accessed 2006].

United States Department of Agriculture (USDA). 2006b. California Agricultural Statistics 2004. National Agricultural Statistics Service, California Field Office publication. Available from URL: http://www.nass.usda.gov/ca [accessed 2006].

United States Department of Agriculture (USDA). 2006c. Transportation and Marketing profile for Merced County. Agricultural Marketing Service. Washington, DC. Available from URL: http://www.ams.usda.gov/statesummaries/ca/county/county_pdf/merced.pdf [accessed 2006].

United States Environmental Protection Agency (USEPA). 2003. NPDES Permit Regulation and Effluent Guidelines for Concentrated Animal Feeding Operations. EPA 821-F-03-003. Office of Water: Washington, DC.

United States Environmental Protection Agency. 2004. National Emission Inventory—Ammonia Emissions from Animal Husbandry Operations. Draft Report. January 30. Available from URL: http://www.epa.gov/ttn/chief/ap42/ch09/related/nh3inventorydraft_jan2004.pdf [accessed 2005].

United States Environmental Protection Agency. 2006. Fine Particle (PM 2.5) Designations. Available from URL: http://www.epa.gov/pmdesignations/regions/region9desig.htm [accessed 2006].

Van Arendonk, J. 1985. Studies on the Replacement Policies in Dairy Cattle. II. Optimum Policy and Influence of Changes in Production and Prices, *Livestock Production Science* 13, 101-121.

VanderSchans, M. 2001. Nitrogen Leaching from Irrigated Dairy Farms in Merced County, California: Case Study and Regional Significance, Unpublished Masters Thesis. Wageningen University, Wageningen, The Netherlands.

Vargas, R., Frate, C., Canevari, W., Campell-Mathews, M., Marsh, B., Klonsky, K., De Moura, R. 2003. 2003 Grain Corn Cost and Return Study – San Joaquin Valley. University of California Cooperative Extension Publication CO-SJ-03. Available from URL: http://coststudies.ucdavis.edu/ [accessed 2005].

Vickner, S., Hoag, D., Frasier, W., and Ascough, J. 1998. A dynamic economic analysis of nitrate leaching in corn production under nonuniform irrigation conditions, *American Journal of Agricultural Economics* 80(May), 397-408.

Ward, M., deKok, T., Levallois, P., Brender, J., Gulis, G., Nolan, B., and VanDerslice, J. 2005. Workgroup Report: Drinking-Water Nitrate and Health--Recent Findings and Research Needs, *Environmental Health Perspectives* 113, 1607-1614.

Wattiaux, M. 2003. Reproduction and Genetic Selection. Chapter 15 in Dairy Essentials. Madison, WI: The Babcock Institute. Available from URL: http://babcock.cals.wisc.edu/downloads/de/15.en.pdf [accessed 2005].

Yadav, S. 1997. Dynamic Optimization of Nitrogen Use When Groundwater Contamination is Internalized at the Standard in the Long Run, *American Journal of Agricultural Economics* 79, 931-945.

Zhao, B. and Chen, S. 2003. Ammonia Volatilization from Dairy Manure under Anaerobic and Aerated Conditions at Different Temperature, Paper # 034148 presented at the 2003 ASAE International Meeting. July 27-30, 2003. Las Vegas, NV.

Table 1. Model Calibration Results.

Quantity	Units	Steady State Value	Comparison Value	Comparison Source[a]
Calves	# of animals	723	517	VanderSchans 2001
Heifers	# of animals	577	308	VanderSchans 2001
Milk cows	# of animals	1445	1731	VanderSchans 2001
Replacement heifers purchased	# of animals	0	--	--
Annualized profit per milk cow ($2005)	$/head	706	1309	Rotz et al. 2003
Ammonia volatilization	kg N/head-yr	41[b]	38	USEPA 2004
			64	Chang et al. 2004
Nitrate leaching	kg N/ha-yr	414	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	mg N/l	895	200-1000	VanderSchans 2001
			500-800	Campbell Mathews 2006
Lagoon inorganic nitrogen concentration	mg N/l	395	300-600	Chang et al. 2005
Applied water (irrigation + pond)	cm/yr	111	124	VanderSchans 2001
Applied chemical (nitrogen) fertilizer	kg N/ha-yr	0	130-280	VanderSchans 2001
Applied solid manure	kg N/ha-yr	0	--	--

[a] VanderSchans 2001 corresponds to comparison values from the Hilmar farm.

[b] Includes heifers and milk cows but not calves. Annual volatilization per milk cow is 57 kg N.

Table 2. Steady-state NMP simulation results without air regulations for various model scenarios and various levels of willingness to accept manure.

WTAM	NPV loss [%]	Milk cows [#]	Leaching [kg N/ha-yr]	Volatilization [kg N/yr]
Scenario 1: baseline parameter values with 412 kg N/ha-yr application limit				
100%	12.3	1,445	6.0	130,569
60%	12.7	1,445	6.0	130,569
20%	14.3	1,445	6.0	130,569
4%	18.8	1,445	6.0	130,568
Scenario 2: improved input management with 412 kg N/ha-yr application limit				
100%	10.3	1,445	8.6	65,834
60%	10.5	1,445	8.6	65,834
20%	11.0	1,445	8.6	65,834
4%	12.7	1,445	8.6	65,834
Scenario 3: uniform irrigation with 1,200 kg N/ha-yr application limit				
100%	6.2	1,445	6.0	132,670
60%	6.3	1,445	6.0	132,670
20%	6.6	1,445	6.0	132,670
4%	7.5	1,445	6.0	132,669
Scenario 4: selective culling with 412 kg N/ha-yr application limit				
100%	10.6	1,392	6.0	130,044
60%	11.0	1,392	6.0	130,044
20%	12.3	1,392	6.0	130,044
4%	16.1	1,392	6.0	130,044

Table 3. Steady-state NMP simulation results with air regulations for various model scenarios and various levels of willingness to accept manure.

WTAM	NPV loss [%]	Milk cows [#]	Leaching [kg N/ha-yr]	Volatilization [kg N/yr]
Scenario 1: baseline parameter values with 412 kg N/ha-yr application limit				
100%	37.6	1,212	5.1	82,463
60%	38.9	1,150	5.2	82,463
20%	41.8	1,036	5.5	82,463
4%	45.1	913	5.9	82,463
Scenario 2: improved input management with 412 kg N/ha-yr application limit				
100%	10.3	1,445	8.6	65,834
60%	10.5	1,445	8.6	65,834
20%	11.0	1,445	8.6	65,834
4%	12.7	1,445	8.6	65,834
Scenario 3: uniform irrigation with 1,200 kg N/ha-yr application limit				
100%	22.1	1,409	4.3	82,463
60%	23.3	1,352	4.4	82,463
20%	26.0	1,241	4.5	82,463
4%	29.1	1,119	4.6	82,463
Scenario 4: selective culling with 412 kg N/ha-yr application limit				
100%	33.4	1,229	5.0	82,358
60%	35.0	1,163	5.1	82,358
20%	38.3	1,035	5.4	82,358
4%	42.0	896	5.9	82,358

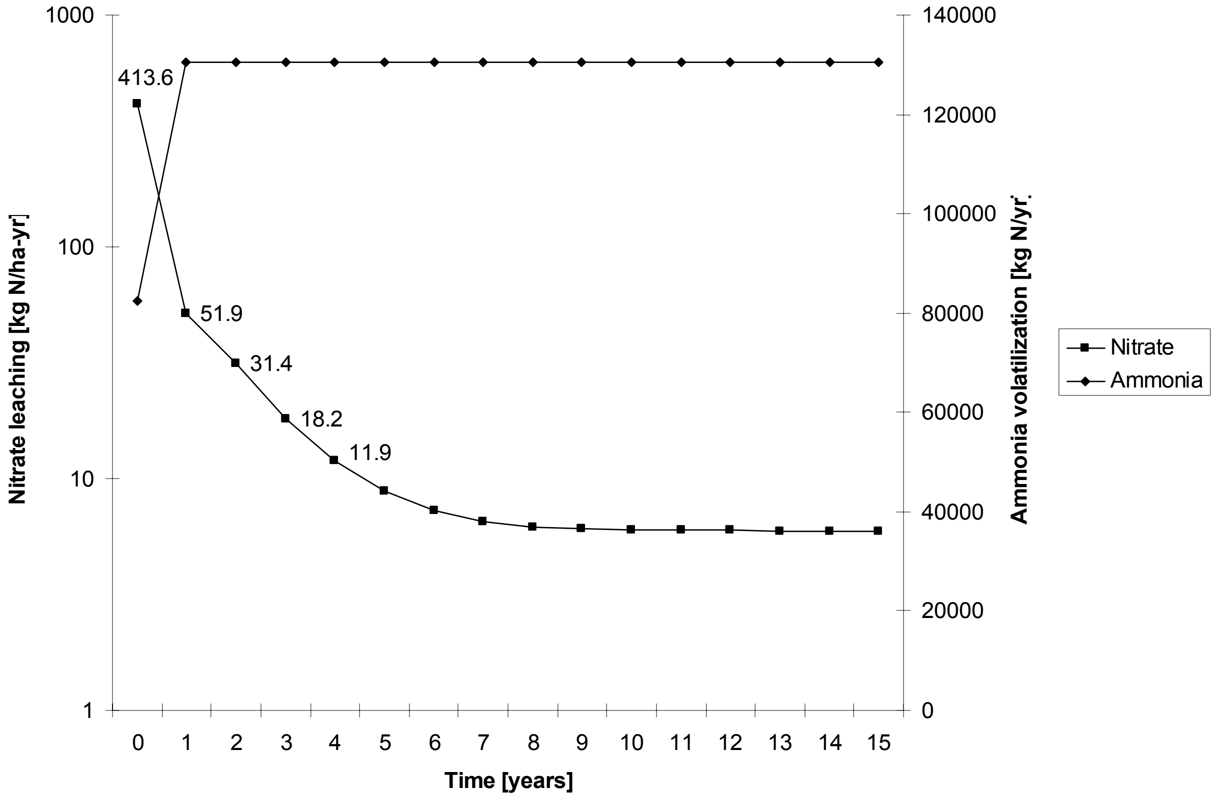

Figure 1: Time paths for nitrate leaching and ammonia volatilization for the baseline scenario without air regulations.