Effects of Nutrient Restrictions on Confined Animal Feeding Operations: Insights from a Structural Dynamic Model

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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 application rates. Policy simulations clarify the importance of dynamic elements and demonstrate three main results: (1) producer cost estimates are significantly higher than previously reported; (2) cross-media pollution effects are potentially quite large; and (3) improved input management appears promising for reducing both emissions and producer 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.

During the past several decades, U.S. livestock industries have continued to consolidate into fewer operations with higher concentrations of animals. From 1965 to 2005, the national average stocking density for hog operations increased from 48 to 912 head per farm. For dairy farms it increased from 13 to 115 head per farm while annual milk production also increased from 3,767 to 8,880 kg per cow (USDA 2006a). Some areas of the country have experienced substantially greater degrees of consolidation than others. As of 2005, hog operations in North Carolina averaged 4083 head per farm and dairy operations in California averaged 763 head per farm with each cow producing 9,709 kg of milk annually (USDA 2006a). Similar trends have occurred in the poultry and beef sectors, as well.

Due to scale economies inherent in livestock production, consolidation has increased farm incomes and enabled operators to continue to meet growing demand for a safe, reliable and inexpensive supply of animal products. But consolidation also has created significant problems regarding management of waste nutrients: whereas the typical animal feeding operation (AFO) in the 1960s was able to dispose of its manure economically on nearby cropland at or below agronomic rates, many modern AFOs produce more waste nutrients than can be utilized locally as fertilizer (Gollehon *et al*, 2001). Currently the most economical solution continues to be over-application of manure, which can produce adverse environmental and health effects as excess nutrients are transported off the farm by natural processes.

Nitrogen and phosphorus emissions from AFOs have received considerable attention from regulators. Both can produce noxious algal blooms and severe fish kills in surface water resources. Nitrate contamination of ground water also continues to be a problem. 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 all water supply wells exceed the standard (Bianchi and Harter 2002). Atmospheric emissions of ammonia-nitrogen also are potentially harmful. Ammonia reacts with oxides of nitrogen and sulfur to form fine airborne particulate matter (regulated as PM2.5 under the Clean Air Act). Currently California's San Joaquin Valley Air District and South Coast Air Basin, both of which contain large numbers of dairy cows, are in non-attainment status for this criteria air pollutant (USEPA 2006).

In response to these problems and to the changing nature of the livestock industry, the United States Environmental Protection Agency (USEPA) recently issued revised guidelines for AFO emissions to surface and ground water (USEPA 2003). State-level agencies that oversee livestock-intensive regions, such as California's Central Valley Regional Water Quality Control Board, San Joaquin Valley Air Pollution Control District, and South Coast Air Quality Management District, also are pursuing more effective regulations for both water and air pollution though typically not in coordination with one another. Although some of these regulations are not yet finalized, nutrient management plans (NMPs) will play an important role in the eventual water quality rulings. NMPs will restrict the rate at which nutrients can be land applied, thereby requiring AFOs with high stocking densities either to change their waste disposal practices or to reduce their herd sizes. If implemented properly, NMPs will significantly decrease the quantity of waste nutrients entering water resources from AFOs but they also may substantially increase operating costs for producers as well as ammonia emissions into the atmosphere (NRC 2002).

These potentially large policy-induced changes in both emissions and farm income have initiated a literature on the anticipated effects of NMP implementation. 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, Magleby and Christiansen (2005) report that dairy farms in the southwest region with lagoon systems would lose 2-4% of net income. 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.[1] Kaplan, Johansson and Peters (2004) estimate that livestock and poultry production could decline by as much as 25% in some regions while increasing in others. And Feinerman, Bosch and Pease (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 basic characteristics (e.g., type of manure handling system), the type of NMP (nitrogen or phosphorus-based), and the amount of off-farm land available for applying manure.

Estimates of the anticipated effects of NMP implementation are obviously useful for policy development. High costs and/or limited or unintended producer responses would suggest that an alternative policy should be pursued; whereas relatively low costs and substantial pollution reductions would strengthen the case for NMPs. Arguably most of the estimates from the existing literature tend to fall into the latter category, as both federal and state regulations continue to move towards NMP requirements in light of these studies. However, there are several reasons to revisit the questions of anticipated producer costs and pollution reductions and the key assumptions and modeling techniques that may be driving these results before widespread implementation of NMPs becomes mandatory.

First, existing farm-level models greatly simplify the AFO management problem. Whereas actual 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 existing literature uses static models that omit these factors and the associated state equations governing their evolution. With fewer constraints imposed on management decisions and outcomes, these models may be underestimating costs and overestimating pollution reductions. Furthermore, static models cannot provide any insights into the temporal aspects of regulations and therefore it is not known how long it will take before anticipated pollution reductions are achieved.[2]

Second, previous estimates are based on average producer characteristics across large geographic regions (covering several states or even the entire U.S.). While knowledge of average costs is useful, it is not obvious that the average cost will apply to many AFOs due to wide and asymmetric distributions of farm characteristics. Regardless of the average implementation cost, the potential range of costs could be quite large, and knowledge of this range (particularly the high end) is no less useful for policy development.

Third, previous studies have focused on hauling and spreading manure off-site as the key producer response to NMP implementation. Although this is a likely response, so too are herd reductions (particularly selective culling of low producing animals), more efficient use of inputs (i.e., conversion of feed nutrients into products rather than waste), and changes to cropping and waste handling systems. With regard to the last item, it is noteworthy that only one previous study examines the possibility of shifting nitrogen emissions from nitrate to ammonia (Aillery et al. 2005), and it predicts at most a small increase in ammonia volatilization if NMPs are implemented without additional air regulations.

In light of these observations, the goals of this study are threefold: (1) to revisit the question of producer cost with a structural model that provides a more accurate representation of 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.

To accomplish these goals, we focus on nitrogen-based NMPs with an application to a large dairy in California's San Joaquin Valley. We consider nitrogen-based NMPs because nitrogen emissions are a significantly more pressing issue in California than phosphorus emissions. This is because the primary mechanism for phosphorus pollution – uncontrolled overland runoff – is highly unusual due to effective regulations as well as to limited rainfall and widespread irrigation in the San Joaquin Valley. However, two key mechanisms for nitrogen pollution – nitrate leaching and ammonia volatilization – are present and continue to degrade environmental quality as described above.[3] We focus on the dairy sector because of its prominence: California currently is the leading dairy state with 19.6% of the nation's cows producing 21.2% of its milk. The farm we model is a typical modern dairy in terms of its size and production technologies. We believe it is representative of an important class of dairy AFOs that are not well characterized by average industry characteristics.[4] We also have excellent data on this dairy that allows us to calibrate our model before conducting policy simulations.

Our findings are summarized as follows. Our estimated NMP implementation costs are two to three times as high as previous estimates for similar AFOs. We find that in addition to off-site hauling of manure, farmers will respond to NMPs by shifting large quantities of nitrogen emissions from nitrate to ammonia if air regulations are not implemented; and they may respond with substantial herd reductions if air regulations are implemented. With regard to short-run effects, we find that although initial reductions in nitrate leaching occur quickly, new steady state levels are not achieved for approximately 7 to 9 years. And of the management options we consider, we find that improved input management potentially has a very large effect on both emissions and implementation costs. We conclude with a discussion of the implications of these findings for both policy and future research.

A Structural Model of a Dairy Farm[5]

Herd Management

Our model farmer works in discrete time and manages a self-replacing herd of calves, heifers and milk cows. 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 $\mathbf{H}$:

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

where $\mathbf{h}_t$ is a ($\bar{a}$ x1) vector representing the number of animals in each cohort during year (*t*); $\boldsymbol{\theta}_t$ is a ($\bar{a}$ 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. Functional forms and parameter values are provided in the appendix.

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. Furthermore, because the marginal contributions of each input to milk, meat and waste outputs are largely unknown, we again follow convention and 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.[6] Table A1 of the appendix 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.

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

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

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. Input and output prices and per-cow quantities as well as the exact specification of the profit function are provided in the appendix (see table A1 and related discussion).

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 nitrogen. 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, 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, solid and liquid wastes are deposited in both the housing structure and the milking parlor and then flushed – with substantial quantities of 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, we specify this type of waste handling system for our model farm.

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. 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 in figure A1 of the appendix. The farmer can then decide how much waste to land apply and how much to export off-site.

Due to differing transportation costs, large dairies often export dried solid waste but retain liquid waste for irrigating and fertilizing crops. However, NMPs will require farmers to significantly reduce their waste application rates. The literature cited herein suggests that farmers are likely to change their waste management practices in response to NMP restrictions by (1) paying to export additional waste from the farm and (2) allowing more liquid waste to volatilize. We incorporate the first response into our model by specifying an off-site waste disposal cost function. Following convention, we assume the dairy is surrounded by a patchwork of land uses with varying suitability for and willingness to receive waste nutrients. The off-site waste disposal cost incurred during crop season (c) and year (t) can be expressed as:

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

where l_{ct} and s_{ct} are the amounts of liquid and solid wastes applied at the dairy, $\mathbf{p}^{d}$ is a vector of unit disposal costs, and $\boldsymbol{\gamma}^{d}$ includes information about the characteristics of the stored waste and the receiving land. 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. Unit costs, parameter values and the exact specification of the cost function are provided in the appendix (see table A2 and related discussion).

We incorporate the second response by allowing the farmer to manipulate the size of the lagoon. We do this 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 (Aillery et al. 2005; NRC 2002). 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 typical 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 of increasing volatilization by resizing the lagoon is representative of the cost 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; however, this management option has not been incorporated into any previous studies.

We therefore specify one additional control variable for the waste handling component of our model, e_t, which is the total surface area of evaporation ponds for liquid waste disposal. Assuming steady-state conditions in the ponds, it is straightforward to derive the rate of ammonia volatilization to the atmosphere from standard physical relationships (Liang, Westerman and Arogo 2002). Table A2 of the technical appendix summarizes these relationships and the parameters 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. 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 (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 $\mathbf{N}$:

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

where $\mathbf{n}_{ct}(\beta)$ 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.[8] 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:

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

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 $g(\beta)$ and multiplying by the total cropped area; aggregate amounts of leaching and volatilization are calculated similarly. The appendix contains the functional forms and parameter values for these relationships.

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

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

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. Again refer to the appendix for parameter values (table A3), control variable constraints, and specific functional forms, including the equations of motion for the state variables.

Optimization

Defining $\pi_t \equiv \pi_t^h + \sum_c \left(\pi_{ct}^{y} - \pi_{ct}^{d}\right)$, 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:

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

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.[9] With this approach, the equations of motion 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 sensitivity analyses and policy simulations. 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 to see 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, our model farm appears to be calibrated quite 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 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 $733/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 $1,284/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 2006b); 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 nearly identical to VanderSchans' best estimate for the Hilmar farm. Corn and wheat yields are high but well 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 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 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 413 kg N/ha-yr to 444 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, which is consistent with the discussion in VanderSchans (2001).

NMP 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 to transition from one steady state to another, what will be the properties of the new steady state, and what will be the economic implications for the dairy. Answers to these questions are needed to evaluate policies for reducing nitrogen emissions. 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.

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, Moffitt and Alt (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.[10]

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 transition path 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 also 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 correspond to 25%, 15%, 5% and 1% of surrounding land that is both suitable for and willing to accept manure.

Scenario 1 in table 2 shows the policy-induced NPV loss and steady state levels for the other variables of concern given our baseline model parameter values. The predicted loss ranges from 12 to 18% of NPV, depending on WTAM. This range is significantly higher than previous estimates of 2-6% for implementing nitrogen-based NMPs at similar livestock operations (Ribaudo et al. 2003, Ribaudo and Agapoff 2005, Huang, Magleby and Christiansen 2005, Aillery et al. 2005). Our estimate includes 2.2% from reduced production (lower crop yields), 4.5% from efforts to increase ammonia volatilization, and 5-11% 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 5 kg N/ha-yr, and the volatilization rate increases from 82,463 to 130,568 kg N/yr. Figure 1 shows that the leaching rate falls precipitously during the first year and then much more gradually thereafter. 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 are adequately captured by the crop production component of the model. However, we will see that culling decisions play a more prominent role when NMPs are implemented in conjunction with ammonia regulations.

Regarding ammonia emissions, our model predicts a 58% increase in volatilization which is substantially more than the "minimal tradeoff" predicted by Aillery et al. (2005). This is largely due to the additional control variable in our model, e_t, which allows the farmer to resize the waste lagoon. 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

A potential weakness of the preceding analysis (and of previous studies) is that it 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 (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. Adopting these practices saves the farmer 2-6% of net income, depending on WTAM, relative to scenario 1. Whether or not these gains would offset adoption costs in the absence of government subsidies is a question we leave for future work; 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 5 to 8 kg N/ha-yr. This increase is due to a combination of several effects. First, because the nitrogen concentration of the waste is lower, 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 waste salts.[11] 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, with perfectly uniform irrigation our model predicts that the NMP constraint could be increased from 412 to 1,175 kg N/ha-yr while still achieving 5 kg N/ha-yr of nitrate leaching. 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 potential weakness of the existing literature is that it does not account for the ability of farmers to selectively cull 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 such culling models do exist (e.g., Van Arendonk 1985), they have not been used in the context of environmental pollution control perhaps due to the difficulty of scaling-up to the farm level a decision model based on individual animal characteristics. However, we can use our model to proxy such culling decisions by introducing cohort milk yield distributions and assuming farmers cull the lowest yielding cows first. To conduct a simple sensitivity analysis, 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. This gives a slightly different unregulated steady state operating position for the farm: profits are 13% higher, the herd contains 1,391 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%. The most interesting result is that 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 NMP-induced increases in ammonia volatilization and the associated air quality problems in livestock-intensive regions, we now consider the likely effects of implementing ammonia regulations in addition to NMP restrictions. Regulations on ammonia emissions could take a variety of forms; as in Aillery et al. (2005), we consider the relatively straightforward case of a quantity restriction. The regulation requires that total ammonia emissions from the farm do not exceed the unregulated steady-state level, but it does not require a reduction below that level. This could be considered a relatively mild restriction, given that air quality regulators in California are actively 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 36-43% 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 the largest portion of the total loss, amounting to 17-35% of net farm income depending on WTAM; in scenario 3 they range from 3-21% of the total. Selective culling again does not have a large effect on costs, and improved irrigation uniformity has a smaller effect than it does in the absence of air regulations.

Discussion and Conclusion

Regarding our first goal – to revisit the question of NMP costs with a structural model that provides a more accurate representation of the dynamic management problem and constraints facing a representative AFO – we find that producer costs estimates are in the range of 12-18% of farm profits for our baseline case. This is significantly higher than previous estimates of 2-6% for implementing nitrogen-based NMPs at similar livestock operations. We do not interpret this result to mean that previous estimates are necessarily wrong; indeed, we make clear that our model differs both in terms of its structure and in terms of the "representative farm" that it considers. Rather we conclude that NMP implementation costs for some producers could be substantially greater than previous estimates. Expected income losses around 15% strike us as qualitatively different from losses around 4%, and could induce unanticipated changes in the industry (e.g., restructuring, additional government cost sharing) or could make policy implementation and enforcement difficult in some regions. Furthermore, the ability of existing subsidy programs (i.e., EQIP) to mitigate these losses is diminished by our analysis because such programs address only the off-site waste disposal cost component; they do not address the additional 6-7% profit loss estimated here. Complicating these issues further is the fact that the impacted producers operate relatively larger farms and produce a large share of the total output. Therefore their operating decisions can have non-trivial effects on markets and local economies. Overall we think NMP implementation will have a considerably larger economic impact than has been suggested by previous farm-level studies.

Regarding our second goal – 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 rates will occur quickly but achieving steady state levels will require 7-9 years. We also predict that ammonia emissions will increase rapidly and that there is considerable risk of substantially degrading air quality in livestock intensive regions if NMPs are implemented without ammonia regulations. This finding is contrary to that of the one previous study that has examined the issue, and it suggests more work is needed to evaluate the nature of the trade-off and what types of measures might be taken to manage it.[12] Issues to consider include the benefits obtained from reducing emissions, including the temporal aspect of exposure to both nitrate and ammonia. That is, ammonia emissions can have an immediate effect on air quality whereas 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 emissions alone do 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.

Regarding our third goal – 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 mixed results. On the one hand, the considerable differences between our estimated costs and cross-media effects versus those of previous studies, as well as our finding that nitrate leaching can actually increase when the waste nitrogen concentration decreases, suggest that structural dynamic modeling 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, but it would be a useful next step. On the other hand, we also find that the dynamics of herd management do not seem to play a large role in 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.

Regarding future policy directions for reducing nutrient pollution from AFOs, this study suggests that the current trend towards mandatory NMP implementation with partial offsets for waste hauling costs provides, at best, a partial solution to the problem. 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 classic 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; but currently such allowances are not being considered.

This study also demonstrates the importance of coordinated air quality regulations and the promising role of improved input management techniques. According to our estimates in table 3, improved input management has the potential to reduce losses by 75% when both NMPs and air regulations are implemented together. However, this finding is based on broad assumptions about currently unproven technologies and the costs producers might incur to adopt them. It also comes with the interesting 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, more work is needed to identify the cost functions for these technologies and then to examine what types of additional incentives – if any – might be appropriate for encouraging their use.

Endnotes

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[1] For the case of NMP implementation without additional air regulations.

[2] 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. This article focuses on dynamic elements of the production process. Linking a model like ours to a dynamic fate and transport model would permit a more complete analysis of the problem.

[3] Nitrates from agriculture also contaminate surface waters directly, and nitrate contaminated ground water also can flow into streams, rivers and lakes via either natural channels or tile drains. The focus of this study is California's Central Valley where most nitrates initially enter ground water via percolation (leaching); therefore we do not consider surface runoff. The scope of this study is limited to initial discharges of nitrogen from farms; therefore we do not consider the eventual fate of nitrates after they enter the ground water system.

[4] The U.S. dairy industry is marked by a large number of very small farms and a small number of vary large farms, with the larger farms producing most of the milk output (NASS 2006).

[5] Due to the level of detail of our model, much of the exposition is contained in the appendix. In this section we present the important variables and relationships that are necessary for understanding our general approach. The reader should consult the appendix for specific functional forms, parameter values, and constraints.

[6] 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.

[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 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.

9 The advantage of using GAMS is that it can readily solve high-dimensional dynamic optimization problems like this one, but it cannot easily incorporate stochastic state equations. 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.

10 The total amount of applied nitrogen in the unregulated steady state is 2195 kg N/ha-yr.

11 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).

12 Readers familiar with Aillery et al. (2005) may wonder why we did not incorporate lagoon covers as in that study, and may conclude this could change our results considerably. However, this is not the case. Aillery et al. consider stricter air regulations than we do that require ammonia emissions to be reduced below the unregulated steady state level. In these cases a lagoon cover may be an optimal response. But for the case we consider it is never optimal to install a cover. We know this because in all cases our model chooses a larger evaporation pond to take advantage of relatively cheap atmospheric disposal. Installing a cover reduces the amount of nitrogen emitted to the atmosphere and increases the amount that must be disposed of by costly off-site hauling.

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Table 1. Model Calibration Results.

Quantity	Units	Steady State Value	Comparison Value	Comparison Source
Calves	# of animals	722	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	733[a]	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	413	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	110	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 $1284. We retain the milk yield parameters reported in table 1 because they are much closer to the reported average for California dairies (USDA 2006b) 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 57 kg N.

[c] Imposing the midpoint of VanderSchans' range (205 kg N/ha-yr) on our model gives a leaching rate of 444 kg N/ha-yr.

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

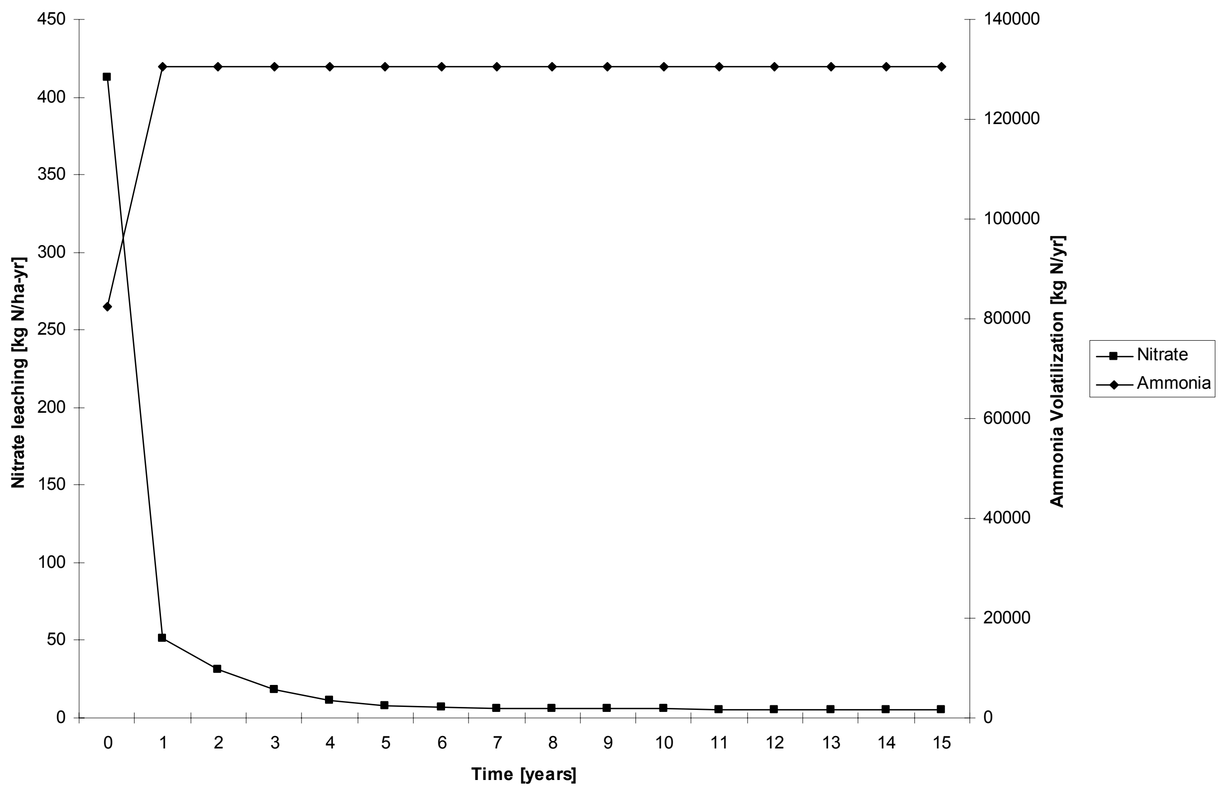
Table 2. Long-run NMP simulation results without air regulations for various model scenarios and various levels of willingness to accept manure.

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%	11.8%	1,445	5	130,568
60%	12.3%	1,445	5	130,568
20%	13.7%	1,445	5	130,568
4%	18.1%	1,445	5	130,568
Scenario 2: improved input management with 412 kg N/ha-yr application limit				
100%	9.9%	1,445	8	65,833
60%	10.1%	1,445	8	65,833
20%	10.6%	1,445	8	65,833
4%	12.2%	1,445	8	65,833
Scenario 3: uniform irrigation with 1,175 kg N/ha-yr application limit				
100%	6.1%	1,445	5	132,603
60%	6.2%	1,445	5	132,603
20%	6.5%	1,445	5	132,603
4%	7.5%	1,445	5	132,603
Scenario 4: selective culling with 412 kg N/ha-yr application limit				
100%	10.2%	1,391	5	130,043
60%	10.6%	1,391	5	130,043
20%	11.8%	1,391	5	130,043
4%	15.5%	1,391	5	130,043

Table 3. Long-run 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%	36.3%	1,184	5	82,463
60%	37.5%	1,141	5	82,463
20%	40.1%	1,029	5	82,463
4%	43.1%	912	5	82,463
Scenario 2: improved input management with 412 kg N/ha-yr application limit				
100%	9.9%	1,445	8	65,833
60%	10.1%	1,445	8	65,833
20%	10.6%	1,445	8	65,833
4%	12.2%	1,445	8	65,833
Scenario 3: uniform irrigation with 1,175 kg N/ha-yr application limit				
100%	21.9%	1,383	4	82,463
60%	23.1%	1,329	4	82,463
20%	25.6%	1,223	4	82,463
4%	28.5%	1,108	4	82,463
Scenario 4: selective culling with 412 kg N/ha-yr application limit				
100%	32.2%	1,214	5	82,358
60%	33.7%	1,151	5	82,358
20%	36.8%	1,025	5	82,358
4%	40.3%	893	5	82,358

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



Appendix A

Herd Management

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:

$$h_{a,t+1} = \begin{cases} \sum_{\tilde{a}\in\{2,3,4\}} \theta_{\tilde{a}t}\gamma^{f}\gamma^{b}_{\tilde{a}}\gamma^{s}_{\tilde{a}}h_{\tilde{a}t}, & a=1 \\ \theta_{a-1,t}\gamma^{s}_{a-1}h_{a-1,t}+\omega_t, & a=2 \\ \theta_{a-1,t}\gamma^{b}_{a-1}\gamma^{s}_{a-1}h_{a-1,t}, & a=3,4,5 \end{cases}, \tag{8}$$

where h_{at} is the number of animals in age cohort (a) during year (t); θ_{at} is the retention rate (1 – the culling rate); γ^{f} is the fraction of calves that are female (we assume bull calves are sold during their first year); 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 $\theta_{5t}\equiv 0$).

Herd Production

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. We also assume that each cow achieves a cohort-specific weight and produces a fixed amount of milk and waste during each lactation. Table A1 provides these cohort-specific quantities, as well as water requirements and fixed operating costs. We also include a herd permit constraint that limits the total number of animal units. Table A1 shows the animal unit value for each age cohort.

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

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

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 A1.

Waste Management

We assume 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) with wastes deposited accordingly. Flows to waste storage are shown in figure A1. Assuming steady-state conditions in the lagoon, the rate of nitrogen flux to the atmosphere (n_t^v) [kg/s] from lagoon disposal is given by a standard physical relationship (Liang, Westerman and Arogo 2002):

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

where K_L [m/s] is the overall mass transfer coefficient for ammonia, F_1 is the fraction of free ammonia concentration in solution, $[TAN]$ [kg N/m^3] is the total ammonia nitrogen concentration, and e_t [m^2] is the area of the lagoon. Table A2 summarizes these and other parameters and relationships used to specify the pond mechanism.

Following Keplinger and Hauck (2006), we specify an off-site waste disposal cost function by assuming the operator must search for available land in the vicinity around the farm. Representing the farm by a circle of radius $\underline{r}$, the area that must be searched can be represented by a disk with larger radius $\overline{r}$ and area given by: $A = \pi\left(\overline{r}^2 - \underline{r}^2\right)$. Following convention we assume disposal costs are a function of distance; therefore we are interested in knowing the average distance that waste must be hauled for disposal. Assuming waste is stored at the center of the stylized farm, the average distance it must be hauled is given by:

(11) $$r^* = \frac{1}{\pi\left(\overline{r}^2 - \underline{r}^2\right)} \int_{\underline{r}}^{\overline{r}} r \cdot 2\pi r \cdot dr = \frac{2\left(\overline{r}^3 - \underline{r}^3\right)}{3\left(\overline{r}^2 - \underline{r}^2\right)}.$$

The area of the farm (99 ha = 0.382 sq mi) is used to calculate $\underline{r} = 0.349$ mi. The radius of the searched area is thus given by: $\overline{r} = \sqrt{\left(A + 0.382\right)/\pi}$. Following Fleming, Babcock and Wang (1998), the area that must be searched is calculated by dividing the quantity of waste nutrients that must be disposed off-site [kg N] by the product of the amount of waste nutrients that can be applied off-site per unit area of land [kg N/ha], the fraction of surrounding land that is suitable for receiving waste [ha/ha], and the fraction of suitable land that is willing to accept manure (WTAM) [ha/ha]. Values for the off-site application rate and the fraction of suitable land are given in table A2.

Assuming revenues can be earned from selling dried solid waste but liquid waste must be shipped off-site at the operator's expense, the waste disposal cost function can be written as:

$$\pi_{ct}^{d} \equiv \left(p^{base} + p^{dist} \cdot r^{*}\right)\left(\overline{l}_{ct} - l_{ct}\right) - p^{sol}\left(\overline{s}_{ct} - s_{ct}\right),$$

where $\overline{l}_{ct}$ and $\overline{s}_{ct}$ are the amounts of liquid and solid waste available for either on-site land application or off-site disposal during each cropping season, l_{ct} and s_{ct} are the amounts applied on-site, p^{sol} is the price received for dried solid waste, p^{base} is the base price for hauling manure off-site, and p^{dist} is the cost per unit distance. The farmer thus affects waste disposal costs by determining how much waste is generated, how much is lost during storage, and how much is applied on-site during each cropping season. The base-plus-distance cost formula is consistent with Fleming, Babcock and Wang (1998) and several other studies. Prices are given in table A2.

Crop Production

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 during crop season (c) and given water infiltration coefficient β. 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:

(12) $$y_{ct}(\beta) = \overline{y}_{c}\left(\frac{1}{1+\alpha_1\left(\alpha_2 + rf_c + \beta\left(i_{ct} + l_{ct}\right)\right)^{\alpha_3}}\right)\left(\frac{1}{1+\alpha_4\left(n_{ct}^{u}(\beta)\right)^{\alpha_5}}\right)$$

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

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

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

Equation (12) specifies the yield (y) [T/ha] of crop (c) at time (t) as a function of the maximum potential yield ($\bar{y}_c$), seasonal rainfall (rf_c) [cm], the water infiltration coefficient β, depth of applied irrigation water (i_{ct}) [cm], depth of applied liquid waste (l_{ct}) [cm], nitrogen uptake rate (n_{ct}^{u}) [kg N/ha], and some parameters α. Equation (13) specifies the nitrogen uptake as a function of the maximum potential uptake ($\bar{n}_c$), seasonal rainfall, the water infiltration coefficient, 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 (14) specifies the amount of plant available nitrogen as a function of the concentration of inorganic soil nitrogen (I_{ct}) [kg N/ha], the water infiltration coefficient, 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_i) [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]. 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). Equation (15) specifies the nitrogen leaching rate as a function of the amount of plant available nitrogen, the water infiltration coefficient, seasonal rainfall, depth of applied irrigation water, depth of applied liquid waste and some parameters.

Following Schwabe and Knapp (2005) and citations therein, we assume the water infiltration coefficient has a log-normal distribution per unit 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 type. Assuming wheat (c') follows corn (c) during each year (t), these equations are given by:

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

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

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

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

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:

$$\pi_{ct}^{y} = \left(p_c^{crop} \sum_j y_{ct}\left(\beta_j\right) F\left(\beta_j\right) - w_c^{fix} - w^i i_{ct} - w^f f_{ct}\right)\left(L - \frac{e_t}{10^4}\right) - w^e \max_{\tau \in \{1 \ldots t\}}\left[e_\tau\right], \quad (20)$$

where p_c^{crop} is the price received for crop (c) [$/T]; $F\left(\beta_j\right)$ is the probability that β is in the interval corresponding to β_j; 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 [$/kg N] and f_{ct} is the amount of chemical fertilizer applied [kg/ha]; L is the total amount of cropland available for crops and ponds; w^e is the annualized cost of constructing evaporation ponds [$/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 he uses less than the total area in the future (e.g., if the herd size is reduced). Table A3 summarizes the parameter values for the cropping system.

Table A1. 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; USDA 2006b
x_{a2}	Wheat silage consumed by cohort (a) [kg/cow-yr]	{861, 2143, 2621, 2621, 2621}	Rotz et al. 1999a; Weiss et al. 1995; USDA 2006b
x_{a3}	Corn grain consumed by cohort (a) [kg/cow-yr]	{522, 102, 3296, 3296, 3296}	Rotz et al. 1999a; Weiss et al. 1995; USDA 2006b
x_{a4}	Soybean meal consumed by cohort (a) [kg/cow-yr]	{0, 0, 13, 13, 13}	Rotz et al. 1999a; USDA 2006b
x_{a5}	Protein mix consumed by cohort (a) [kg/cow-yr]	{0, 0, 151, 151, 151}	Rotz et al. 1999a; USDA 2006b
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; Wattiaux 1999; Chang et al. 2004
--	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; Wattiaux 1999; Chang et al. 2004
p^{milk}	Price received for milk [$/kg]	0.310	USDA 2006b
p_a^{herd}	Price received for culling cohort (*a*) [$/animal]	{353, 838, 633, 633, 633}	USDA 2006b; Wattiaux 1999; Chang et al. 2004
p^{repl}	Price paid for replacement heifers [$/cow]	1500	USDA 2006b
$\{w_1 \ldots w_5\}$	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 A2. Manure Storage and Disposal Specifications.

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]	$2.229\times10^{-6}\exp\left(0.236\cdot wind\right)$	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
--	Off-site manure application rate [kg N/ha]	170	Kellogg et al. 2000
--	Fraction of land suitable	25%	Kellogg et al. 2000;

	for receiving manure		USDA 2006c
p^{sol}	Price received for dried solid manure [$/kg N]	0.14[a]	Norwood, Luter and Massey 2005; Vargas et al. 2003
p^{base}	Base cost for manure hauling [$/gal]	0.00764	Fleming, Babcock and Wang 1998
p^{dist}	Distance cost for manure hauling [$/gal-mi]	0.00329	Fleming, Babcock and Wang 1998

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

[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 A3. Crop Production Parameters.

Symbol	Description	Value	Source
rf_c	Rainfall during growing seasons for corn, wheat [cm]	{10, 25}	WRCC 2006
$\overline{y}_c$	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
$\overline{n}_c$	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. 2004
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 [$/T]	{117, 116}	Vargas et al. 2003; Brittan et al. 2004
w_c^{fix}	Fixed crop production cost for corn, wheat [$/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[b]	Moser et al. 1998
ρ	Discount factor	0.9615[c]	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 exported manure is used as fertilizer. Other uses may be possible (e.g., energy generation) depending on local conditions.

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

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

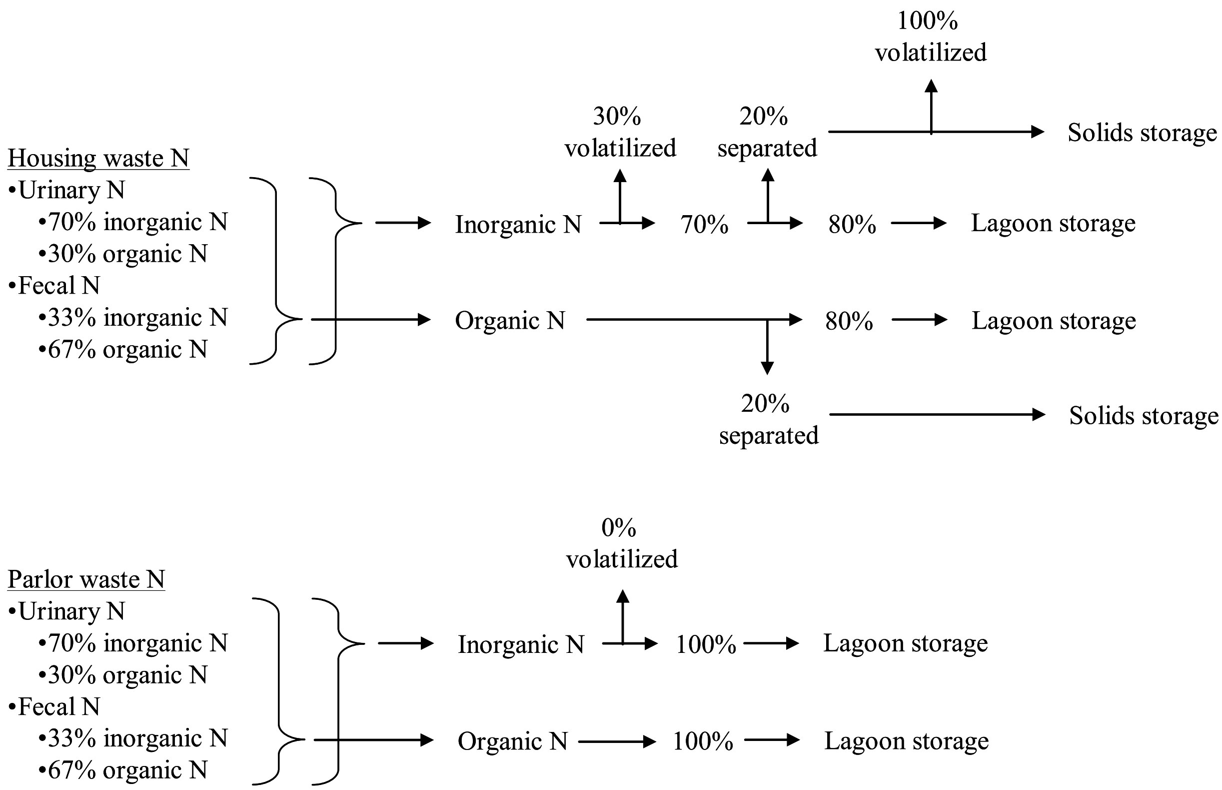


Figure A1: Predetermined waste nitrogen flows (Chang et al. 2004; VanderSchans 2001).