Evaluating Pollution Control Policies Using a Farm-level Dynamic Model: An Application to Large Dairy Farms in California

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Abstract

Animal waste from concentrated animal feeding operations (CAFOs) is a significant contributor to the nitrate contamination of groundwater. Some manures also contain heavy metals and salts that may build up either in cropland or groundwater. To find cost-effective policies for pollution reduction at the farm level, an environmental-economic modeling framework for representative CAFOs is developed, where the owner of the operation is a profit-maximizer subject to environmental regulations. The model incorporates various components such as herd management, manure handling system, crop rotation, water sources, irrigation system, waste disposal options, and pollutant emissions. Decision rules from the optimization problem demonstrate best management practices for CAFOs to improve their economic and environmental performance. Results from policy simulations suggest that direct quantity restrictions of emission or incentive-based emission policies such as a field emission tax are much more cost-effective than the standard approach of limiting the amount of animal waste that may be applied to fields. Furthermore, incentive-based emission policies are shown to have advantages over direct quantity restrictions under certain conditions. We also demonstrate the importance of taking into account the integrated effects of water, nitrogen, and salinity on crop yield and nitrate leaching as well as the spatial heterogeneity of nitrogen/water application when designing policy mechanisms.

Keywords: pollution control, policy mechanism, animal waste, crop production, nitrate, groundwater, dynamic optimization

1. Introduction

The growing world population, together with globally converging diets, has fueled the sustained rise in demand for food of animal origin. Between 1964-66 and 1997-99, the human population roughly doubled, while the number of domestic animals tripled (FAO, 2003). In the U.S., the national average stocking density for dairy operations increased from 57 to 139 head per farm from 1992 to 2009 (USDA, 2010). As shown in Figure 1.1, the situation is particularly

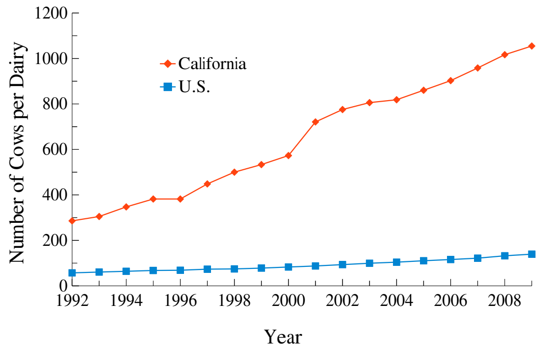

Figure 1.1: Average number of cows per dairy in U.S. and in California

noticeable in California. California has been the nation's leading dairy state since 1993. As of 2009, the average size of a dairy herd in the state was 1055 cows, much higher than the national average level (CDFA, 2010). For Kern County, one of the five leading dairy counties in the state, the average number of cows in a dairy operation is up to 3190 (CDFA, 2010). Higher farm incomes due to economies of scale will sustain the trend toward larger and more concentrated animal feeding operations (CAFOs).

Another significant change throughout the world is land use transformation. For the U.S. agricultural sector specifically, changes have taken place in cropping patterns with the total amount of crop land relatively stable (Lubowski et al., 2006). In California, more than 1.2 million acres of land for field crops has been converted to vineyards, vegetables, and orchards in the past three decades (Cooley et al., 2009). Consolidation combined with the deceasing acreage for field crops lead to less land available for animal waste disposal. In addition, animal waste (especially dairy and swine manure) is costly to move relative to its nutrient value. Therefore, the common practice of operators continues to be over-application of animal waste on land close to the facility. Excess nutrients transported off the farm can produce adverse environmental and health effects.

Nitrogen and phosphorus emissions from CAFOs have received considerable attention from regulators. Either nutrient can accelerate algae production in receiving aquatic ecosystems leading to potentially large algal blooms and a variety of problems including clogged pipelines, fish kills, and reduced recreational opportunities (USEPA, 2000). Furthermore, nitrate-nitrogen in groundwater is a potential threat to public health. Two medical conditions have been linked to excessive concentration of nitrate in drinking water: blue-baby syndrome in infants, and stomach cancer in adults (Addiscott, 1996, Fleming and Adams, 1997). The U.S. Geological Survey reported in 2009 that nitrate was the most common pollutant derived from man-made sources that had higher concentrations than human-health benchmarks (DeSimone, 2009). High levels of nitrates are found most frequently in aquifers underneath agricultural regions, such as the basin-fill aquifers in the Southwest and the Central Valley aquifer system in California (DeSimone, 2009). Nitrate contamination of the groundwater is therefore the main focus of this study. We leave other potential contaminants from CAFOs such as pathogens, antibiotics, and air pollutants to future research.

The United States Department of Agriculture (USDA) and Environmental Protection Agency (USEPA) have endeavored to control the emissions from AFOs since the late 1990s. The early regulations addressed the vast majority (about 95 percent) of AFOs by voluntary programs including environmental education, locally led conservation, financial assistance, and technical assistance (USDA and USEPA, 1999). In response to the increasingly severe problem of nutrient pollution, USEPA published a new rule for CAFOs in 2003. One of the key components is nutrient management plans (NMPs). Each CAFO is required to prepare and implement a site-specific NMP for animal waste applied to land (USEPA, 2003). Based on this rule, the California Regional Water Quality Control Board (CRWQCB) published a General Order for dairies in 2007. As of 2012, the land application rate of nitrogen in the Central Valley will typically be limited to 1.4 times the agronomic rate of crop nitrogen removal (CRWQCB, 2007). If implemented properly, NMPs will significantly decrease nitrogen run-off and leaching. However, developing and implementing such a plan may substantially increase operating costs for producers.

The evaluation of the economic impacts for CAFOs to comply with the NMP requirement has received significant attention in the literature. Ribaudo et al. (2003) evaluate the costs for CAFOs to meet a nutrient standard at the farm, regional, and national levels. They use a simulation model developed by Fleming et al. (1998). The model has two components: cost of transporting and spreading manure to a specific amount of receiving land, and benefits from replacing commercial fertilizer with manure nutrients. Their farm-level analysis suggested a 0.5-2.0 percent increase in production costs for large dairies when the willing-to-accept-manure (WTAM) by surrounding crop producers is 20 percent (Ribaudo et al., 2003). When competition for spreadable cropland is introduced in the regional analysis, the costs increase to 40-50 percent of the total net returns, not including the offset by savings from replacing commercial fertilizers (Ribaudo et al., 2003). Kaplan et al. (2004) utilize a sector model to evaluate regional adjustments of productions and prices when CAFOs meet nutrient standards. Whether the secondary price effects are sufficient to offset the compliance costs depends on crop producers' WTAM. An unanticipated result in their study is the increase of nitrogen leaching in some areas due to the expanded cropland acres and changes in crop production. Huang et al. (2005) report that 6-17 percent of medium and large dairy farms in the southwest US would suffer from the NMP requirement while other dairies in the region could avoid income loss by leasing additional nearby cropland at the current cash rent, which may be tenuous assumption. Two recent studies use Geographic Information Systems to improve the modeling of spatial transportation of manure at the regional level (Aillery et al., 2009, Paudel et al., 2009).

Although these studies provide a full perspective on potential economic impacts for CAFOs to meet nutrient standards, their models are static and fail to reflect changes in management practices other than spreading manure on additional land and changing cropping patterns. Baerenklau et al. (2008) implement a structural dynamic whole-farm model, including herd management, crop production with non-uniform irrigation, waste disposal, and cross-media effects of nitrogen pollution (via nitrate leaching and ammonia volatilization). The results indicate the profit losses due to NMP could be much greater than previously anticipated, even without allowing for regional competition for land. They point out that regulating leaching rates rather than nitrogen application rates would be more cost-effective. They also suggest modeling endogenous irrigation system choice, given the observed potential benefit of more uniform irrigation.

The problem of over-application of animal waste to land is a classic agricultural nonpoint source pollution problem. For crop agriculture, various policies have been studied at the field and farm level to control nutrient pollution from over-applying commercial fertilizers, including both command-and-control regulations (e.g., restrictions on nitrogen or water use) and incentive-based instruments (e.g., nitrogen use tax, effluent tax on nitrogen percolation, tax on nitrogen surplus, water tax, cost sharing for adoption of modern irrigation technologies). A tax on nitrogen use and a control on fertilizer application have been shown to be the least effective policies (Wu et al., 1994, Helfand and House, 1995, Berntsen et al., 2003), suggesting the NMP requirement for CAFOs might not be the most cost-effective policy to reduce pollution. However, except for Baerenklau et al. (2008), few studies undertake field-level and farm-level analyses of controlling nutrient emissions from CAFOs. This may due to the fact that animal waste also contains high concentrations of salts and there is very limited information on how crop yields and leaching rates respond to varying application rates of water, nitrogen, and salts.

Similar to Baerenklau et al. (2008), this paper use a dynamic environmental-economic model to investigate cost-effective policies for nitrate pollution control at the farm level. The model departs from and builds upon this work by incorporating soil salinity as a new state variable, as well as the selection of manure handling systems, irrigation systems, crop rotations, and water sources as new control variables. Furthermore, novel functional relationships derived from simulated data account for changes in crop yields and leaching rates due to varying levels of water, salinity, and nitrogen in the soil.The model is validated with data from representative California dairies and is utilized to simulate the effects of NMPs and alternative polices, including direct quantity restrictions of emission and incentive-based emission policies. Our results suggest that a direct quantity restriction of field emission or a field emission tax is much more cost-effective than the NMP requirement. Furthermore, emission tax is shown to have advantages over direct quantity restrictions under certain conditions. We also demonstrate the importance of taking into account the integrated effects of water, nitrogen, and salinity on crop yield and nitrate leaching as well as the spatial heterogeneity of nitrogen/water application when designing policy mechanisms.

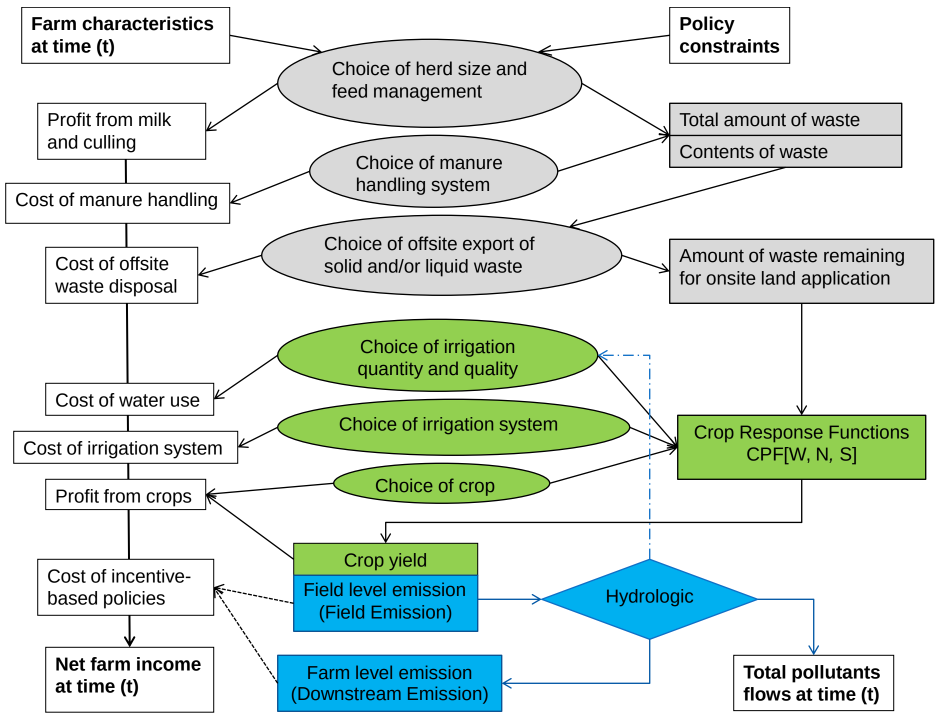

Figure 2.1: Key elements of the Integrated Animal-Crop Operation Model

2. The Integrated Animal-Crop Operation Model

Our integrated animal-crop operation model is adapted and expanded from the model of Baerenklau et al. (2008). Following their approach, Figure 2.1 summarizes the key inputs and outputs (bold text), choice variables (ovals), and sub-components (shaded). Although the model is calibrated for large dairy farms, it can be easily adapted to other AFOs. In the subsections below we describe the three main building blocks that make up the model: animal model, crop model, and economic model.

2.1. Animal Model

2.1.1. Herd Growth and Production

The animal model is comprised of a livestock production model coupled with a herd growth model. The livestock production model calculates the annual output levels of milk, meat and waste from animal characteristics, such as herd size, herd composition, feed, and management. The herd growth model traces the livestock population over time, depending on the calving rate, the mortality rate, the culling rate and the purchasing rate.

Baerenklau et al. (2008) simulate herd dynamics but find it to be not as important as soil nitrogen dynamics. Following their suggestion, the herd growth component of our model does not include the formal transition equations for each age cohort. Instead, we only trace the total number of animals on farm, assuming the structure of the herd is fixed (i.e., the numbers of calves, heifers, and milk cows increase or decrease proportionally when the operator buys or sells animals). The herd dynamics are thus simplified by reducing the number of state variables to one.

For the production levels of milk and meat, we follow convention and assume the feed consumption, weight, and milk production is fixed for each age cohort. However, unlike poultry and swine farms for which both the waste mass (e.g., the amount of nitrogen in kilograms) and the waste volume (e.g., the amount of wastewater in gallons) per animal are usually relatively constant, the waste volume generated by a dairy farm significantly depends on the its management practices, particularly the manure handling system.

2.1.2. Waste Management

The animal operation produces a waste slurry. Some waste solids are separated and sold off-site as fertilizer. The remaining liquid waste needs to be disposed, usually in one of the three ways: land application, wetland treatment, and anaerobic digestion (Morse et al., 1996, Schaafsma et al., 1999, Paudel et al., 2009). The characteristics of the dairy manure in California, its bulk and relatively low primary nitrogen and phosphate levels, generally make it infeasible for most dairies to participate in a centralized digester or a constructed wetland treatment (Hurley et al., 2007). Therefore we assume in the model that animal waste is applied to croplands either on-site or off-site.

Average water use in a dairy is 95-175 gallons per cow per day, depending on how much water is used to flush manure from the milking parlor and bedding facility (Bray et al., 2011). The volume of liquid wastewater is equal to the total water use less water in milk and evaporative losses from the production system. The characteristics of the waste, together with the required hauling distance, will determine the volume of waste exported and the associated cost. An increase in total waste volume will increase the transportation cost of waste disposal. Following convention, the distance hauled is a function of available land suitable for spreading animal waste and the WTAM of nearby land owners. See Ribaudo et al. (2003) and Baerenklau et al. (2008) for a detailed discussion.

Two common manure handling systems are considered in the study: flush-lagoon and scrape-tank. The scrape-tank system is labor and capital intensive but use much less water per cow and thus produces a smaller volume of waste. The two are also different in the method of on-site waste spreading. Under the flush-lagoon system, wastewater shares the same pipelines with the irrigation system. Therefore, the non-uniformity of an irrigation system determines the non-uniform land application of animal waste. Under the scrape-tank system, waste is transported and spread to land via tractors so we assume it can be uniformly applied over the field. Currently the flush-lagoon system is used in about two-thirds of all the California dairies and typically employed in the Central Valley (Hurley et al., 2006). The annual total cost of a manure system equals the annualized fixed cost plus annual operating costs (i.e., power costs and labor costs) and the cost from non-drinking water consumption. The cost is \$47 /cow-yr for flush-lagoon system and \$121 /cow-yr for scrape-tank system (Bennett et al., 1994), while the water demand of flush-lagoon system is $241.77\,\mathrm{m}^3$/cow-yr and that of a scrape-tank system is $131.24\,\mathrm{m}^3$/cow-yr (Bray et al., 2011).

See Appendix B for the details and the mathematical formulation of the animal model.

2.2. Crop Model

2.2.1. Crop Choice

Typical cropping systems for California dairies consist of sequential winter forages and summer corn rotation. Manure is usually diluted with irrigation water to avoid applying high concentrations of salts to fields, a practice that would diminish crop yields. Greater dilution tends to flush more nitrogen into the underlying aquifer. This suggests that nitrogen-hungry, salt-tolerant crops, as well as more uniform irrigation systems (i.e., systems that reduce over-watering parts of a field and thus minimize flushing chemicals through the soil) could be a promising cost-effective strategy for pollution reduction. The operator can use either (i) more nitrogen-hungry crops to reduce the amount of nitrogen in the root zone that is often flushed to groundwater and/or (ii) more salt tolerant crops to reduce soil flushing thereby reducing the amount of water that passes through the root zone and carries nitrogen and salts to groundwater. However, none of the crop yield and emission functions in the literature has taken into account the effects of interactions and feedback mechanisms in the whole plant-water-nitrogen-salinity system. The existing functions only account for and compute partial effects, such as plant-water, plant-water-salinity, and plant-water-nitrogen (Dinar et al., 1986, Pang et al., 1997, Knapp and Schwabe, 2008). Therefore this study utilizes software developed for simulating water flow and solute transport in variably saturated porous media as well as root water and nutrient uptake (HYDRUS-1D) to generate simulated crop datasets, from which both yield and emission functions are developed. We generate datasets for a subset of crops which appear to be economically beneficial for the operator, require large amounts of nitrogen, and can withstand high salt concentrations. See Appendix D for the estimation of these crop response functions.

Several studies incorporate cropping patterns by assuming the farmer chooses from a set of candidate crops and allocates a proportion of available land to each selected crop at the beginning of each year, either in linear programming models or in dynamic frameworks (Wu et al., 1994, Haouari and Azaiez, 2001, Knapp and Baerenklau, 2006). Linear programming models have also been used to simulate crop rotation (El-Nazer and McCarl, 1986, Detlefsen, 2004). The basic concept is that the acreage for a crop in next year should not be larger than the acreage for the previous crop in the sequence. Note that the optimal crop rotation from an agronomic perspective can be different from the optimal crop rotation from an economic perspective, similar to the case of fisheries management where the maximum sustainable yield is different from the optimum sustainable yield. An ideal model would include the dynamics of soil characteristics (i.e., the carry-over of moisture, nitrogen, and salts) and allow the choice of alternative crops at the beginning of each season. Two recent studies consider crop rotation in a dynamic optimization framework. Livingston et al. (2008) set up crop choice as a dynamic optimization problem and solve it over an infinite time horizon. They assume expected crop revenues (also crop yields) depend on current and the previous year's crop choices as well as current fertilizer applications. Under a similar corn-soybean rotation, Cai et al. (2011) examine both one-year and two-year carry-over effects. In their model, crop yields are assumed to be exogenous with the yield level responding to current and previous planting decisions. In both studies, the carry-over effects on crop yields are characterized by a very short history of crop choices, which are captured in the coefficients of simple regression analyzes of experimental crop yields. The dynamics of soil characteristics and the full effects of carry-over mechanisms have not been explored in literature.

There are two possible ways of implementing a crop rotation over multiple years: 1) plant the entire farm to a single crop each year, and 2) divide the farm into equal parts, and rotate the crops within each parcel in such a way that the total acreage of each crop grown on the farm is about constant each year (Hazell and Norton, 1986). The latter approach is the common practice in the real world because it can maximize the utilization of some machines and equipment, provide continuous feed for animals, and reduce risk by growing a portfolio of crops rather than a single crop. The former involves fewer state and choice variables and thus would make the whole optimization problem easier to solve. Over long-term planning horizons, the two approaches are equivalent. Here we model the crop system following the first approach.

Crop production costs are from University of California Cooperative Extension and prices are from National Agricultural Statistics Service.

2.2.2. Irrigation System

Improved irrigation uniformity has been shown to be a promising method of cost reduction under environmental regulations. Following Knapp and Schwabe (2008), the spatial heterogeneity of water distribution over the field is represented by a water infiltration coefficient β, which has a log-normal distribution with mean $E[\beta] = 1$ and standard deviation $SD[\beta]$. Data on common irrigation systems is readily available from previous studies. $SD[\beta]$ can be calculated from the Christiansen uniformity coefficient of each system. See Table C.1 for irrigation system data and the calculation of $SD[\beta]$.

To make this model tractable, the log-normal distribution of β is discretized into three intervals. These intervals can be interpreted as subareas of the field with different water infiltration coefficients β_i, $i \in \{1, 2, 3\}$. We do this in a way such that $[\beta_1, \beta_2, \beta_3] = [0.3, 0.9, 1.7]$ and characterize the three types of subareas as under-irrigation field, average irrigation field, and over-irrigation field. See Table C.2 for discretized intervals over the field.

Initially three types of irrigation systems are considered in this study: 1/2-mile furrow, 1/4-mile furrow, and linear move. According to Baerenklau et al.

(2008), 1/2-mile furrow is the common system used by dairy farms in the Central Valley. Our preliminary results show 1/2-mile furrow would never be adopted and 1/4-mile furrow system is the optimal choice under the baseline scenario. Personal communications with farm advisers in Tulare County (Carol Frate, Oct 21, 2011) suggest that 1/4-mile furrow system is the most common one in that county, while 1/2-mile furrow is popular in the northern part of the valley where more surface water is available. Since our empirical study focuses on the area of Tulare and surrounding counties, we only include 1/4-mile furrow and linear move system in the simulations and analyzes reported here.

Our analysis does not consider the scheduling of irrigation and fertilizer application. We assume the operator follows the recommended scheduling for a given crop and irrigation system.

2.2.3. Other Specifications

Climate and Soil. We collect data on the region including Tulare County and Kern County, which accommodate more than 1/3 of the cows in the state. The climate data (temperature, precipitation, pan evaporation) are ten-year averages from 2000-2009 over four CIMIS stations. The main soil type of farmland in Tulare & Kern is sandy loam/ loamy sand (USDA, 2007): bulk density 1.5 g/cc (range $1.35-1.70$ g/cc), saturated hydraulic conductivity 28 μm/ sec (range $14-42$ μm/ sec) (USDA, 2009). According to the soil survey in USDA (2009), the highest soil salinity in this region is around 20 dS/m, which is well covered by my selected intervals of soil salinity for the crop response functions. The dynamics of moisture and salinity is adapted from Bresler (1967) and Knapp (1984). The dynamics of nitrogen closely follows Baerenklau et al. (2008).

See Appendix C.1 for balance relationships and C.2 for dynamics of water, nitrogen, and salts in soil.

Fertilizer Source. Commercial fertilizers and noncommercial fertilizers (animal waste) can be applied to maintain proper nutrition for the crops. Three sources of nitrogen fertilizer in our model: liquid waste, solid waste, and commercial fertilizer at a cost of \$0.59/kgN.

Water Source. The farmer can import high quality surface water ($EC = 0.15$ dS/m and $N = 1$ mg/l) with a price of \$2.58/ha-cm (Vargas et al., 2003), pump deep groundwater ($EC = 1.18$ dS/m and $N = 10$ mg/l) at a cost of \$8.84/ha-cm (Marques et al., 2003), and pump shallow groundwater through capture wells at a cost of \$2.58/ha-cm (Schans, 2001). The quality of groundwater is updated from GeoTracker GAMA, an online database provided by the California State Water Resources Control Board.

Surface water prices vary among irrigation districts in the Central Valley. Depending on which irrigation district the farm belongs to and the groundwater depth in that region, the average water cost can range from \$1.62/ha-cm to \$11.35/ha-cm (Hutmacher et al., 2003).

In this paper, water quality is fixed. In a subsequent paper, a hydrologic model will be coupled with the current model to make the groundwater quality endogenous and take into account reductions in downstream groundwater emissions due to wastewater recycling via supply wells, and then field emission control polices and downstream emission control polices will be compared and analyzed.

2.3. Economic Model

Net farm income equals the net revenues from herd production, waste disposal, and crop production less the environmental policy costs.

$$\pi_t = \pi_t^{herd} + \pi_t^{waste} + \pi_t^{crop} - \pi_t^{policy}$$

The objective function is:

$$\max_{\{\theta_{t,g}, l_{t,k}, s_{t,k}, sw_{t,k}, gw_{t,k}, fl_{t,k}, M, I, R\}} \left[\sum_{t=1}^{T} \left[\eta^t \pi_t \right] + \eta^T \mathbf{p}^{\mathbf{herd}} \left(\boldsymbol{\zeta}_1 h_{T,G} \right)^{\top} \right] \quad (2.1)$$

subject to transition equations, mass balance requirements, non-negativity constraints, herd permit limits, and policy-related constraints.

The model has 10 state variables: 1 for herd size and 9 for three soil characteristics (soil organic nitrogen, soil inorganic nitrogen, and soil salinity) across three types of field subareas. Currently the stochastic nature of parameters such as milk price and crop price is not included in the model, so rather than set up the problem in a dynamic programming framework, we treat it in *Mathematica* as a constrained non-linear programming problem due to the high dimensionality. The model has three discrete choice variables: manure handling system M, irrigation system I, and crop rotation R. The choice of manure system can be modeled in a similar way to the choice of irrigation system, which is a classical problem of technology choice. Modeling crop rotation under spatial heterogeneity is a challenge. The acreage-allocation model is not feasible here, because it is difficult to track the crops grown at each field type when the acreage for a crop can be freely assigned each year. This may be why we see in the literature acreage-allocation models with multiple crops and spatial dynamic models with a single crop but not spatial dynamic models with multiple crops (or crop rotation). We circumvent the problem by assuming the operator commits to a manure handling system, irrigation system, and crop rotation at the beginning of the planning horizon without switch in future. At the beginning of each year in the planning period, the operator decides how many cows ($\theta_{t,g}$) to buy or sell and the amount of liquid animal waste ($l_{t,k}$), solid animal waste ($s_{t,k}$), surface water ($sw_{t,k}$), groundwater ($gw_{t,k}$), and commercial fertilizers ($fl_{t,k}$) to be applied to the field for each season (k) within that year. η corresponds to a discount rate of 4% in 2005 dollars. The last term of Equation 2.1 represents the salvage value of the herd. See Appendix A-D for details of the terms in the objective function.

There are two ways to mathematically model discrete variables: 1) mixed integer programming, and 2) activity analysis. Mixed integer nonlinear programming problems are difficult to solve, especially for large scale optimization problems. We already have a high dimensional model with several nonlinearities, so we choose the method of activity analysis. An integral component required for the activity analysis is a "technology-cost" matrix which describes the inputs and outputs as well as associated costs for each management practice(Duraiappah, 2003). In this sense, crop rotation can also be modeled via activity analysis, since each pattern of crop rotation can be viewed as an alternative farming technology. We use the KNITRO package to solve the constrained non-linear optimization problem under alternative activities (i.e., alternative combinations of management practices), among which the one with the highest net farm income is the optimal solution of the whole dynamic optimization problem (in the following sections we denote the activities associated with the optimal solution "the optimal activities").

3. Baseline Simulation and Discussions

We first utilize a sequential solution procedure (using the solution of one single-period model as the initial values for the subsequent single-period model) to establish a feasible solution for the dynamic problem, and then use it as the starting point to solve for the long-term optimal dynamic solution under the baseline scenario.The planning horizon is 30 years (i.e., 60 seasons). All the simulated scenarios can reach a steady state over the first 24 years, with boundary effects for some scenarios in the last 6 years (1 rotation-year)[1]. Therefore, the following analyses are based on the results of the first 24 years. Table 3.1 compares the steady state values of the baseline scenario against available data. Animal numbers are similar to those in the Hilmar site. As Baerenklau et al. (2008) point out, the difference in annual profit per cow is due to different assumptions of milk production. The field emission of nitrogen is low compared to that reported in Schans (2001), which is probably because I assume a deeper root zone of 3 meters. In summary, my model appears to be calibrated well.

With no environmental regulations in the baseline scenario, the operator optimally selects flush-lagoon as the manure handling system, 1/4-mile furrow as the irrigation system, and corn-wheat as the crop rotation. The herd size remains steady through time, constrained by the herd permit. Figure 3.1 displays the optimal path of soil organic nitrogen, soil inorganic nitrogen, and soil salinity, which vary depending on the field type (i.e., under-irrigation, mean-irrigation, or over-irrigation).[2] The optimal decision rule for seasonal irrigation,

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[1]Here I use the term "steady state" in a broader sense than just a terminology associated with infinite deterministic dynamic programming problems.

[2]For soil inorganic nitrogen, under-irrigation $<$ mean-irrigation $<$ over-irrigation; for soil salinity, under-irrigation $>$ mean-irrigation $>$ over-irrigation. This is because the concentration of inorganic nitrogen is much higher than the concentration of salts in animal waste. Meanwhile, the amount of nitrates and salts that can be carried through the soil is limited during each irrigation and thus during the whole season. Therefore, leaching significantly affects the total amount of salts in soil but not the total amount of inorganic nitrogen. More nitrogen

Table 3.1: Model Validation

Variables	Units	Steady State Value	Comparison Value	Comparison Source
Calves	# of animals	723	517	Schans (2001)
Heifers	# of animals	578	308	Schans (2001)
Milk cows	# of animals	1445	1731	Schans (2001)
Heifers purchased	# of animals	0	0	–
Annualized profit	$/cow	757.2	1309	Rotz et al. (2003)
Field Emission (Nitrate leaching)	kgN/ha-yr	158.3	202-660	Schans (2001)
Downstream Emission	kgN/ha-yr	129.8	–	
Total applied water	cm/yr	150.4	124	Schans (2001)
Applied surface water	cm/yr	116.8	–	
Applied groundwater	cm/yr	0	–	
Recycled drainage water	cm/yr	0	–	
Applied commercial fertilizer	kgN/ha-yr	0	130-280	Schans (2001)
Applied liquid waste	kgN/ha-yr	2304.8	–	
Applied solid waste	kgN/ha-yr	0	–	
Irrigation system		1/4-mile furrow	1/4-mile furrow 1/2-mile furrow	Personal communication (Carol Frate, farm adviser of Tulare County)
Manure handling system		flush-lagoon	flush-lagoon	Hurley et al. (2007)
Crop rotation		corn-wheat	corn-small grains	Crohn et al. (2009)

as shown in Figure 3.2, suggests that in order to maintain a certain level of salinity, the operator periodically applies large volumes of high quality water to flush the salts out of the soil. This leads to the cyclical patterns in the paths of soil inorganic nitrogen and soil salinity. We do not see a similar pattern for soil organic nitrogen, since water is the transporting medium of dissolved salts (including inorganic nitrogen) but not organic nitrogen.

Figure 3.3 depicts the optimal decision rule for fertilizer application. In the baseline scenario, the operator does not apply commercial fertilizer or solid waste on site. In reality, farmers are usually concerned about certain risks associated with manure fertilizer, such as pathogens and weeds and the fact that organic nitrogen is not immediately plant-available. Therefore they also use some commercial fertilizer. This is why we see the difference between our simulated value and the comparison value for applied fertilizer in Table 3.1. We do not consider these issues for the dairy operator but for surrounding land owners we use three levels of WTAM (20%, 60%, and 100%) to account for these concerns and perform sensitivity analysis. The results shown in this section are for the WTAM level of 60%. Furthermore we assume 25% of surrounding land is suitable for spreading animal waste (Baerenklau et al., 2008).

Figure 3.4 shows the water infiltration and nitrate leaching for each field type. Flushing of salts also carries more nitrates from the root zone to groundwater. The effect is only and especially significant for the mean-irrigated field type. For the under-irrigated field type, there is no excess water even during flushing. For the over-irrigated field type, there is enough excess water to carry all leachable nitrates through the soil even if there is no flushing.

Table 3.2 summarizes the total available water, crop relative yield, and field emission of nitrogen for each field type over the planning horizon. Although flushing significantly increases the leaching for the mean-irrigated subfield, the main contribution of field emission is from the over-irrigated subfield, due to the high non-uniformity of the 1/4-mile furrow system. The over-irrigated field type makes up 18.28% of the field (Table C.2), gets 30.83% of total irrigation, produces 19.85% of total crop yield, but accounts for 77.19% of total field emission of nitrogen.

To further illustrate the effects of non-uniform irrigation, we report similar information in Table 3.3 from the optimization results under an alternative activity where the linear move system is adopted instead of furrow. Compared to the optimization results under the optimal activity, the amount of applied water decreases by 5.63%, but the total relative yield increases by 2.53% and the total amount of nitrogen field emission decreases by 45.58%. Figure 3.5 displays the optimal paths of soil inorganic nitrogen, soil salinity, and field emission for each field type under this alternative activity, as well as the decision rules for irrigation and fertilizer application. With the linear move irrigation system,

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accumulates in the field type which receives more animal waste, and more salts accumulate in the field type which receives less irrigation water.

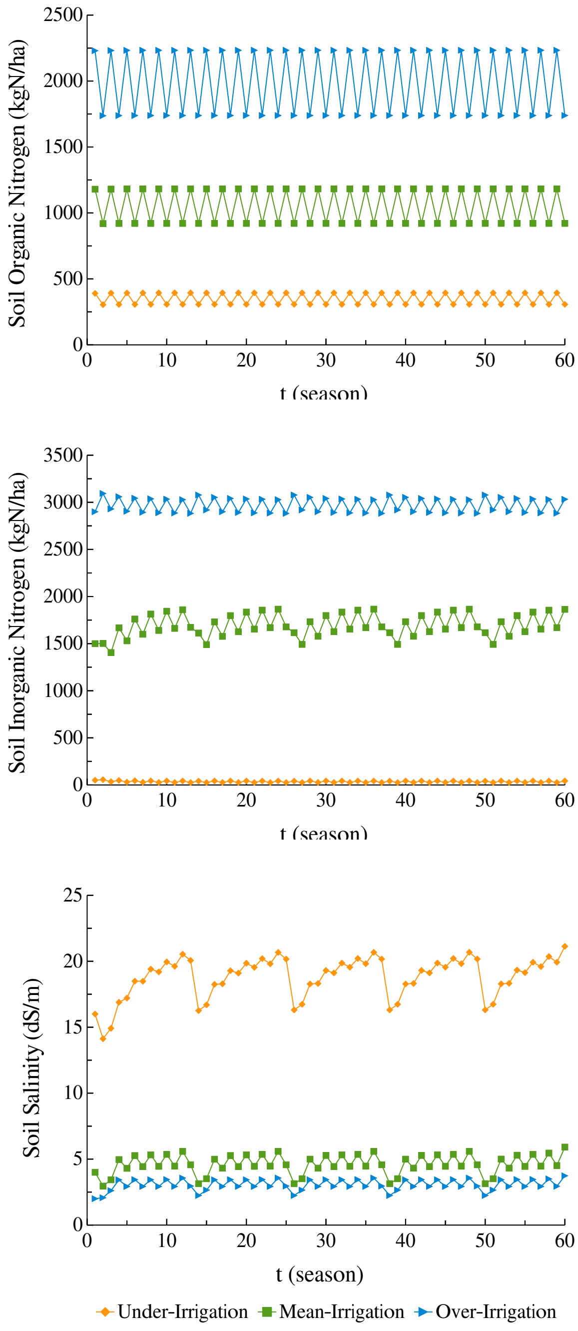

Figure 3.1: Baseline: paths of soil organic nitrogen, soil inorganic nitrogen, and soil salinity for each field type under the optimal activities (flush-lagoon, 1/4-mile furrow, corn-wheat rotation)

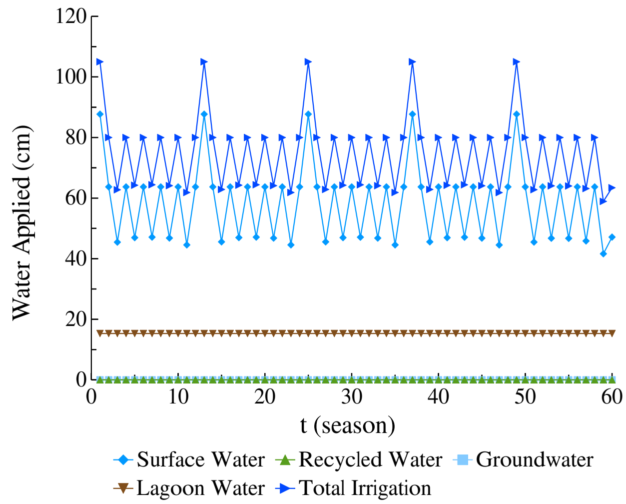

Figure 3.2: Baseline: decision rule for irrigation under the optimal activities (flush-lagoon, 1/4-mile furrow, corn-wheat rotation), given three sources of irrigation water

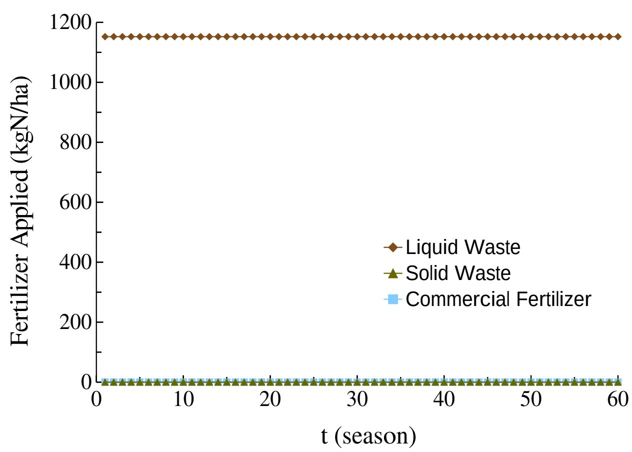

Figure 3.3: Baseline: decision rule for fertilizer application under the optimal activities (flush-lagoon, 1/4-mile furrow, corn-wheat rotation), given three sources of fertilizer

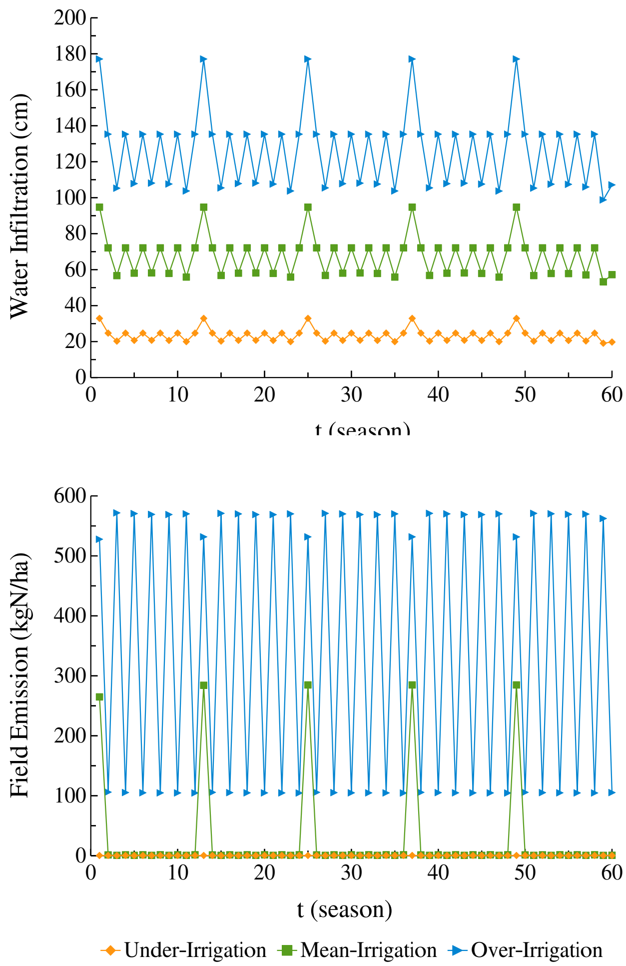

Figure 3.4: Baseline: paths of water infiltration and nitrate leaching for each field type under the optimal activities (flush-lagoon, 1/4-mile furrow, corn-wheat rotation)

Table 3.2: Baseline: irrigation, relative yield, and field emission of nitrogen for each field type under the optimal activities (flush-lagoon, 1/4-mile furrow, corn-wheat rotation)

	Field Type			
	Under-Irrigation	Mean-Irrigation	Over-Irrigation	Total
Irrigation	80.81	2415.80	1112.69	3609.29
(cm)	[2.24%]	[66.93%]	[30.83%]	[100.00%]
Relative Yield	1.51	24.70	6.49	32.69
	[4.61%]	[75.54%]	[19.85%]	[100.00%]
Field Emission	0.97	865.55	2932.81	3799.33
(kgN/ha)	[0.03%]	[22.78%]	[77.19%]	[100.00%]

Table 3.3: Baseline: irrigation, relative yield, and field emission for each field type under an alternative combination of activities (flush-lagoon, linear move, corn-wheat rotation)

	Field Type			
	Under-Irrigation	Mean-Irrigation	Over-Irrigation	Total
Irrigation	0.03	2688.53	717.66	3406.22 (*-5.63%*)
(cm)	[0.00%]	[78.93%]	[21.07%]	[100.00%]
Relative Yield	0.00	29.16	4.36	33.52 (*+2.53%*)
	[0.00%]	[86.99%]	[13.01%]	[100.00%]
Field Emission	0.00	44.97	2022.47	2067.44 (*-45.58%*)
(kgN/ha)	[0.00%]	[2.18%]	[97.82%]	[100.00%]

Table 3.4: Baseline : irrigation, relative yield, and field emission for each field type under an alternative combination of activities (scrape-tank, 1/4-mile furrow, corn-wheat rotation)

	Field Type			
	Under-Irrigation	Mean-Irrigation	Over-Irrigation	Total
Irrigation	99.36	2399.41	1039.82	3538.59 (*-1.96%*)
(cm)	[2.81%]	[67.81%]	[29.39%]	[100.00%]
Relative Yield	1.55	24.46	6.53	32.54 (*-0.47%*)
	[4.78%]	[75.16%]	[20.06%]	[100.00%]
Field Emission	3.03	1042.49	1491.93	2537.45 (*-33.21%*)
(kgN/ha)	[0.12%]	[41.08%]	[58.80%]	[100.00%]

the operator stops periodically applying large volumes of high quality water. The linear move system is more uniform than the furrow system and thus can maintain the soil salinity at certain levels for the subfields without flushing. Therefore, the amount of nitrate emitted from the mean-irrigated subfield is greatly reduced from 865.55 kgN/ha to 44.97 kgN/ha, a 94.80% decrease. Also, nitrate leaching from the over-irrigated subfield decreases around 31.04% because of the improved uniformity of water and waste distribution. The net farm income is lower under this activity though due to the higher cost of the linear move system. This implies that a relative simple policy of subsidizing more uniform irrigation systems might be able to achieve a substantial reduction in field emission.

A switch from the flush-lagoon system to the scrape-tank system can also effectively reduce nitrate leaching, but through different mechanisms. Figure 3.6 displays some optimal paths and decision rules under this alternative activity.The over-irrigated subfield has the smallest amount of both nitrogen and salts in soil because it has the highest level of leaching and because I assume animal waste is uniformly applied while mixed irrigation water is not. For the same reasons, the steady state level of soil salinity for the over-irrigated subfield is lower than that under the optimal activity (Figure 3.1). Similarly, the steady state level of soil salinity for the under-irrigated subfield is higher than that under the optimal activity. Compared to the results under the optimal activity, the amount of nitrate leaching from the over-irrigated field type significantly decreases under this alternative waste management activity, as shown in Table 3.4 . The mean-irrigated field type now contributes over 40% of the total amount of nitrate leaching, which suggests that salt flushing has significant effects on nitrate leaching under uniform fertilizer application and non-uniform irrigation. This is also of great importance to crop agriculture, where commercial fertilizers are usually uniformly applied but water infiltrations varies spatially. For the whole field, the amount of applied water, the total relative yield, and the total amount of field emission decreases by 1.96%, 0.47% and 33.21% respectively, compared to the results under the optimal activity. Again, the net farm income

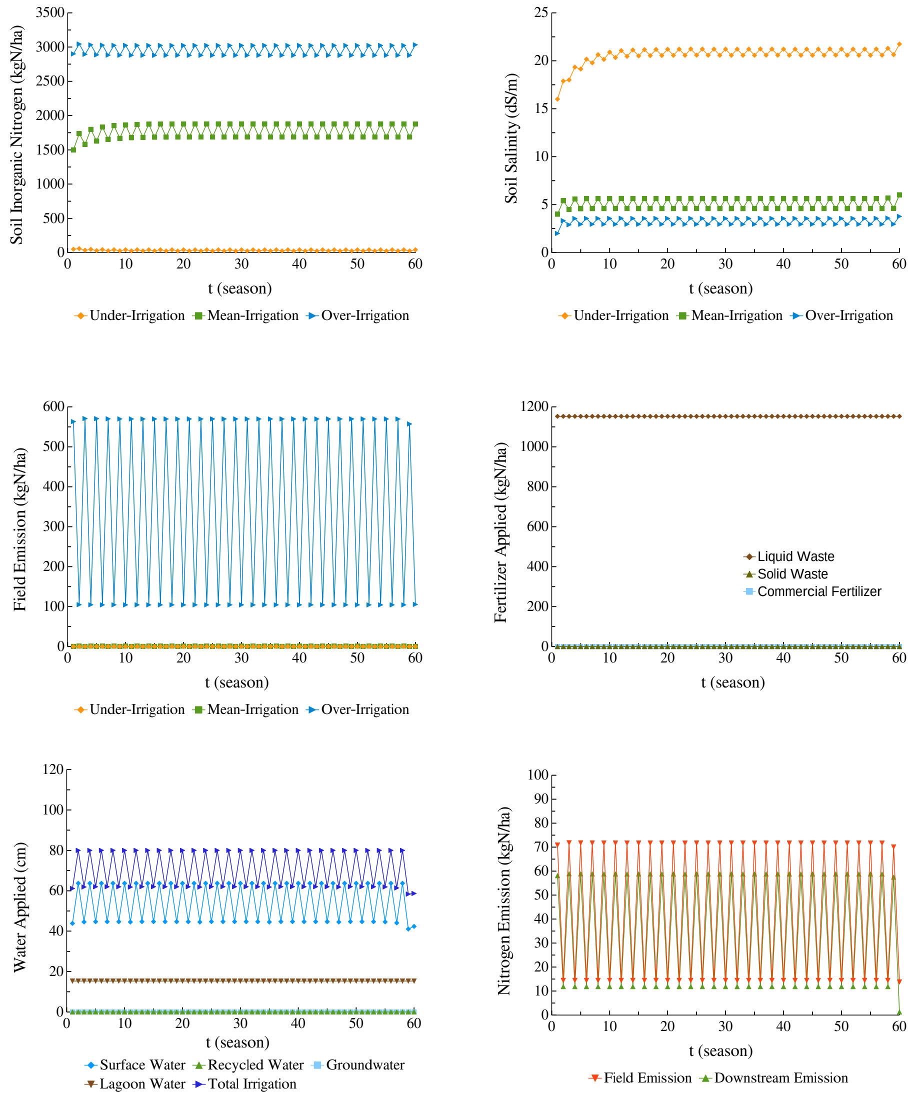

Figure 3.5: Baseline: paths (soil inorganic nitrogen, soil salinity, field emission) for each field type, decision rules (irrigation, fertilizer application), and seasonal nitrogen emissions under an alternative combination of activities (flush-lagoon, linear move, corn-wheat rotation)

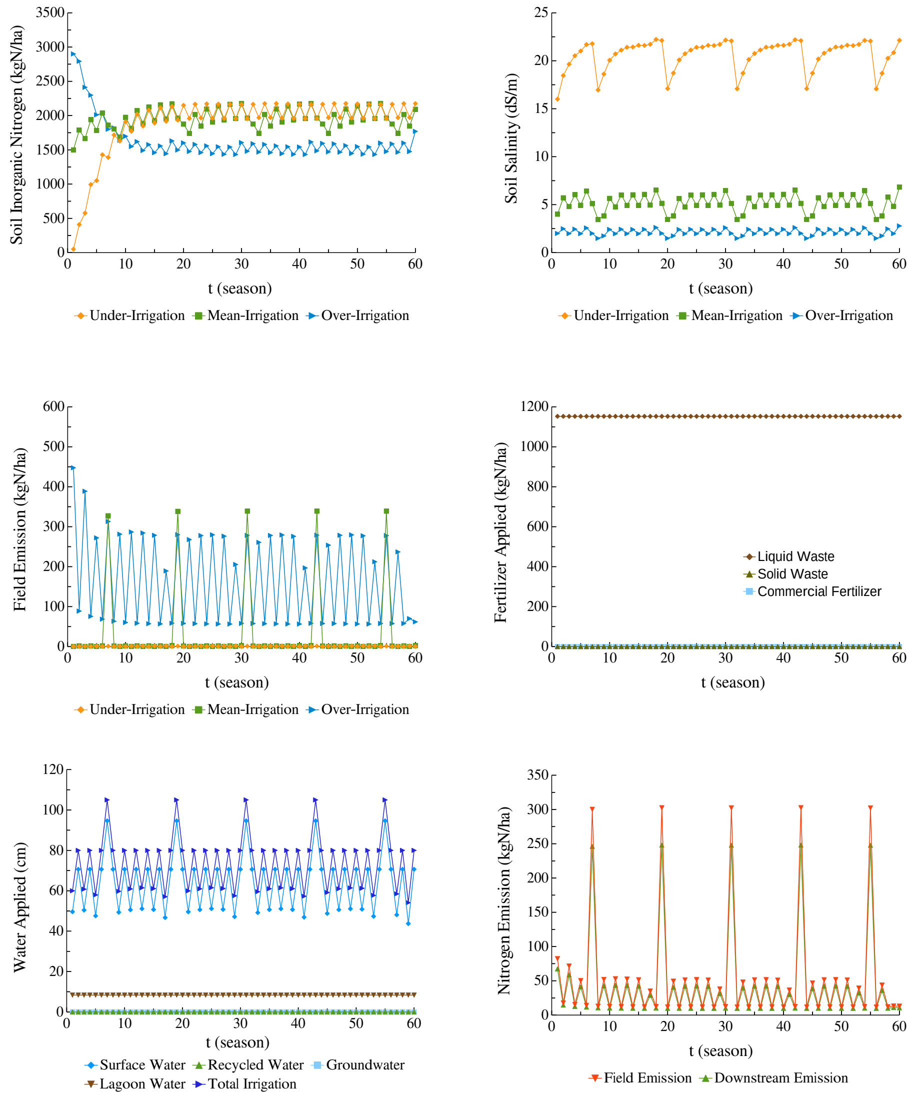

Figure 3.6: Baseline: paths (soil inorganic nitrogen, soil salinity, field emission) for each field type, decision rules (irrigation, fertilizer application), and seasonal nitrogen emissions under an alternative combination of activities (scrape-tank, 1/4-mile furrow, corn-wheat rotation)

is lower under this activity due to the higher cost of the scrape-tank system, but a policy that subsidizes water-saving manure collecting systems and/or more uniform waste distribution systems might also be able to achieve a substantial reduction in field emission.

4. Policy Simulations and Discussions

4.1. Nutrient Management Plans

According to CRWQCB (2007), potential nitrogen sources available for each crop should at least include "manure, process wastewater, irrigation water, commercial fertilizers, soil, and previous crops". In our model, manure and process wastewater are equivalent to the animal waste, and nitrogen from previous crops is captured by soil dynamics. NMPs do not distinguish between organic nitrogen and inorganic nitrogen, either in dairy waste or in soil. Following convention we assume only inorganic nitrogen is available for crops and thus is restricted by the NMP requirement, which is included in the model as a constraint on seasonal availability of inorganic nitrogen.

We assume the operation is initially at the steady state derived from the baseline scenario and then solve for the dynamically optimal practices under the NMP constraints. Results show that the operator optimally selects scrape-tank as the manure handling system, 1/4-mile furrow as the irrigation system, and corn-wheat as the crop rotation under the NMP scenario. The main results are displayed in Figure 4.1 . Due to the NMP constraint, the operator hauls almost half of the liquid waste off site. This results in the significant decreases in soil organic nitrogen, soil inorganic nitrogen, and soil salinity levels of the mean-irrigated and over-irrigated subfields, even compared to the baseline results under the same activity (Figure 3.6). Another change in the management practices is the irrigation pattern. Although the total amount of surface water applied over the planning horizon increases little, the water is smoothly applied without flushing. That is why the soil salinity of the under-irrigated subfield remains high. Compared to the baseline scenario under its optimal activities, both the field emission and the downstream emission of nitrogen decrease by 84.10%. Total crop revenue increases by 6.24% but the operator still suffers a heavy net income loss of 27.40%, primarily due to the high cost of offsite waste hauling.

4.2. Field Emission Limit

Our policy simulations are designed to evaluate the cost-effectiveness of alternative policies compared to NMPs. Therefore, we use the nitrogen leaching level of the optimal solution under NMPs as a reference point. The total amount of field emission over the 24 years horizon is 604.2 kgN/ha, or approximately 25.2 kgN/ha per year, which is set as the annual cap of field emission[3].

[3] I also test a 6-year cap of 151.05 kgN/ha for field emission. The optimal solution is similar to that under the annual cap, with the net farm profit slightly higher due to the added

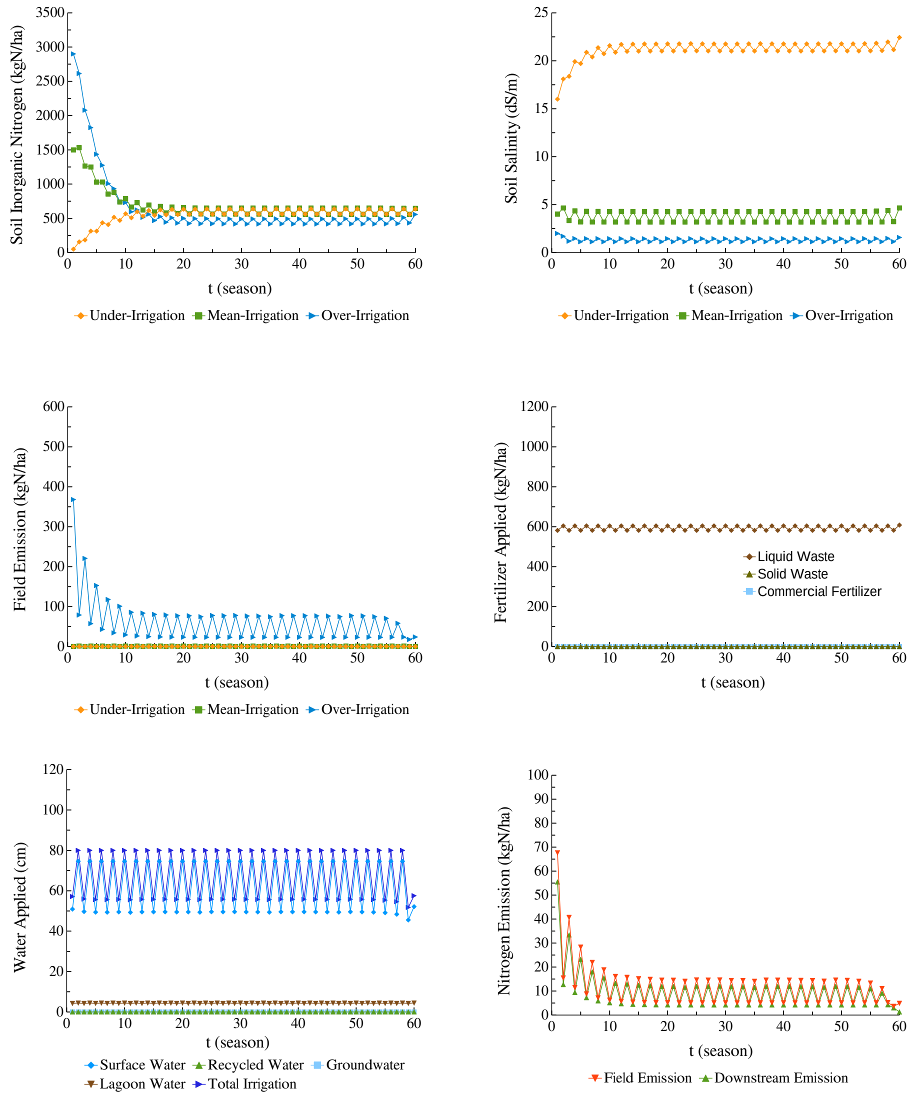

Figure 4.1: Nutrient Management Plans: paths (soil inorganic nitrogen, soil salinity, field emission) for each field type, decision rules (irrigation, fertilizer application), and seasonal nitrogen emissions under the optimal activities (scrape-tank, 1/4-mile furrow, corn-wheat rotation)

The optimal activities under the FEL scenario are flush-lagoon as the manure handling system, linear move as the irrigation system, and corn-wheat as the crop rotation. Figure 4.2 displays the main results. Unlike under the NMPs scenario, the operator does not transport any liquid waste off site with a field emission limit. Instead, the operator controls the rate of field emission by applying less irrigation water and thus holding a large pool of nitrogen in the soil. Denitrification in the unsaturated zone can transform the total available inorganic nitrogen into nitrogen gas at a rate of λ_k, which is a fixed parameter in the model. If more inorganic nitrogen remains in the soil over the season, more becomes nitrogen gas and less nitrate is leached. Therefore, rather than lets nitrate and salt leached down to the aquifer, the operator takes advantage of the natural processes to reduce the field emission of nitrogen.

The reduction in irrigation mainly happens in summer, since the winter crop is more salt-tolerant and under the baseline scenario field emission from the summer cropping is five times more than that from the winter cropping (Figure 3.5). Less irrigation leads to higher levels of soil salinity in the subfields, which can reduce th crop yields. Compared to the optimal results of the baseline scenario, net farm income decreases by only 0.79%, with 7.20% of crop yields sacrificed to meet the field emission standard. This implies that quantity control of intermediate pollution is much more cost-effective than quantity control of polluting inputs for the case of nitrogen emission. The reasons are twofold: First, there is limited information on the relationships between certain inputs and the pollution, especially in the case of multiple inputs. Secondly, and what is more important, the field emission limit creates incentives for the operator to examine the contribution of other management practices to pollution in addition to land application of waste, such as the choice of irrigation system and the pattern/rates of irrigation, which can affect the natural attenuation of nitrogen.

4.3. Field Emission Charge

A per-unit effluent charge can be applied to field emission. For each combination of activities, we derive the field emission charge that would produce the same amount of field emission as NMPs and FEL. A lump-sum return of emission charge does not alter marginal conditions so the optimal activity of FEC (or DEC) when the charge revenue is returned to the industry is same as that when the charge revenue is not returned. At an emission charge of $2.5/kgN/ha, the operator achieves the same level of emission reduction at a net income loss of 0.79% (emission charge is not counted as production cost under the assumption that it would be returned as a lump-sum). The optimal activity and other management practices are same as that under the field emission limit.

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flexibility provided by this longer time horizon. In practice, emission limit or charge would be imposed on estimated (or modeled) nitrogen emissions.

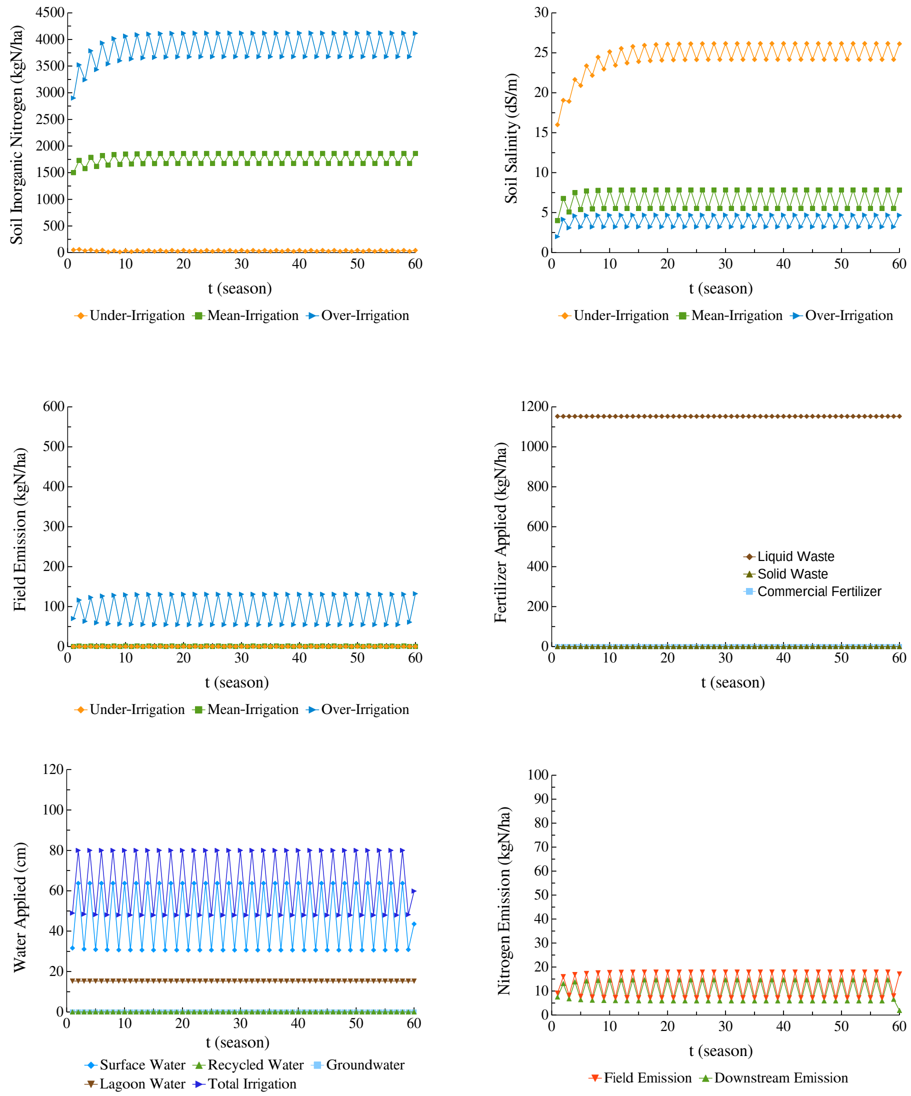

Figure 4.2: Field emission limit: paths (soil inorganic nitrogen, soil salinity, field emission) for each field type, decision rules (irrigation, fertilizer application), and seasonal nitrogen emissions under the optimal activities (flush-lagoon, linear move, corn-wheat rotation)

4.4. Surface Water Tax

We propose a surface water tax as an alternative policy for two reasons: 1) previous studies (Wu et al., 1994, Helfand and House, 1995) show that a water tax might be an effective second-best policy to deal with nutrient pollutions, because water is the carrying medium, 2) a water tax is relatively straightforward to implement with relatively low transaction costs. Similar to the emission tax, we attempted to derive the surface water tax that would produce the same amount of nitrate leaching as NMPs. However, with groundwater as a substitute for surface water, the decrease in nitrate leaching corresponding to the increase in surface water tax is discontinuous. Therefore, we derive the water tax that would produce no more nitrogen leaching than that under the NMP scenario.

Due to the discontinuity in nitrate leaching, it is difficult to set a reference level to evaluate alternative combinations of activities or to exactly quantify the relative cost-effectiveness of different policies. However, it is noteworthy that under the activities of flush-lagoon, linear move, and corn-wheat rotation, the operator achieves greater leaching reduction with a lower profit loss under the surface water tax scenario than under the NMP scenario. Figure 4.3 and Figure 4.4 demonstrate how the optimal results change as the surface water tax increases. There are two tipping points associated with the surface water tax: 1) the tax rate above which the operator stops using imported surface water to flush salts, and 2) the tax rate above which the operator purely relies on groundwater pumping. The total amount of nitrate leaching smoothly decreases as the surface water tax increases to three times the price of surface water which is \$2.58 /cm-ha, but at that point nitrate leaching remains twice as high as under the NMP scenario. As the water tax passes this tipping point, the operator stops flushing salts and the nitrate leaching decreases at an increasing rate. When the surface water tax reaches 4.33 times the price of surface water (the second tipping point), there is a sudden and significant decrease in the amount of nitrate leaching to 24% of that under the NMP scenario. The sudden substitution between imported surface water and groundwater is likely due to the significant difference in water quality. The operator does not rely on groundwater until the total cost of surface water is 2.75 times the cost of pumping groundwater, implying a high value of water quality.

In summary, the ordering of policies in terms of cost-effectiveness in the case of field nitrogen emissions would be field emission charge/field emission limit, surface water tax, and NMPs.

5. Conclusion

This study use a dynamic environmental-economic model to empirically evaluate alternative policy mechanisms for pollution control at the field and the farm level. The optimized characteristics of the animal-crop operation are consistent with available data. Novel crop response functions are derived from simulated data to account for the effects of interactions and feedback mechanisms in the whole plant-water-nitrogen-salinity system. The spatial heterogeneity of water

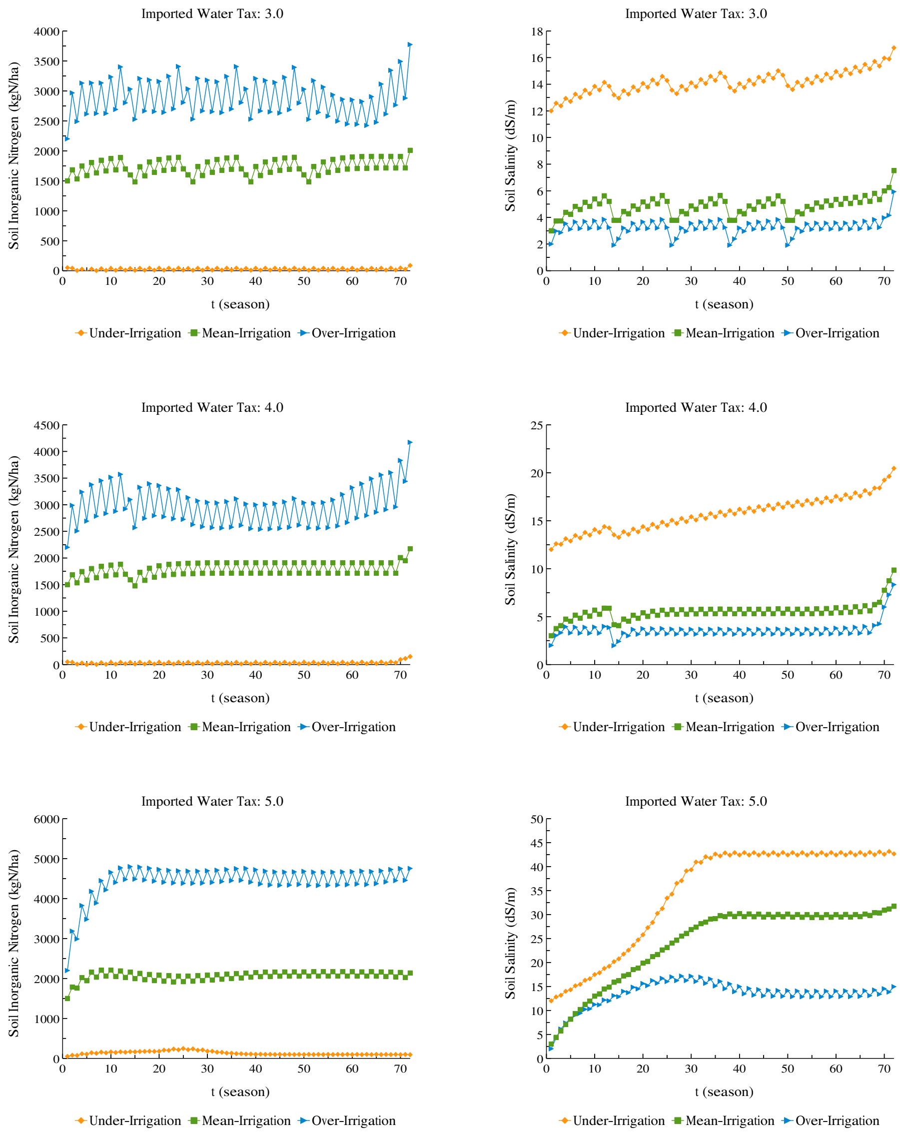

Figure 4.3: Imported water tax (3, 4, and 5 times the price of surface water): paths of soil inorganic nitrogen and soil salinity for each field type under a specified combination of activities (flush-lagoon, linear move, corn-wheat rotation)

Figure 4.4: Imported water tax (3, 4, and 5 times the price of surface water): decision rule for irrigation and path of nitrate leaching for each field type under a specified combination of activities (flush-lagoon, linear move, corn-wheat rotation)

and nitrogen application over the field, combined with the integrated effects of water, nitrogen, and salinity on crop yield and nitrate leaching, has been shown to have significant effects on both the pattern and the total amount of field emission. Modeling of the temporal and spatial dynamics of soil characteristics is necessary to account for these factors and should be incorporated in future research.

Simulated results from the baseline scenario and various policy scenarios suggest that direct quantity restrictions of emission or incentive-based emission policies are more cost-effective than the standard approach of limiting the amount of animal waste that may be applied to fields. Policies targeting pollution creates incentives for the operator to examine the effects of other management practices to reduce pollution in addition to controlling polluting inputs.

In the next step of the research, a hydrologic model will be coupled with the current model to make the groundwater quality endogenous and take into account the reductions in downstream groundwater emissions due to wastewater recycling via onsite supply wells. Field emission control polices and downstream emission control polices will then be compared and analyzed.

AppendixA. List of Index Symbols

$t = 1, \ldots, T,$ rotation-years

$g = 1, \ldots, G,$ years in a rotation-year. G depends on crop rotation duration. $1\ rotation{-}year = G\ (Julian)\ years.$

$k = 1, \ldots, K,$ seasons in a rotation-year. To simulate double cropping per year, $K = 2G$. $k = 1, 3, \ldots, 2G-1$, summer season; $k = 2, 4, \ldots, 2G$, winter season.

$j = 1, \ldots, J,$ number of intervals to represent the spatial heterogeneity of water infiltration over the field, given irrigation system I. Each interval has its own water infiltration coefficient β_j^I.

$I \in \{I_1, I_2\},$ irrigation system type. I_1 is the ¼-Mile Furrow system and I_2 is the linear move system.

$M \in \{M_1, M_2\},$ manure handling system type. M_1 is the flush-lagoon system and M_2 is the scrape-tank system.

$R \in \{R_1, R_2\},$ crop rotation. R_1 is corn-wheat rotation and R_2 is cotton-wheat rotation.

AppendixB. Animal Model

AppendixB.1. Herd Dynamics

The operator works in discrete time and manages a herd of calves, heifers, and milk cows. Each year the operator decides how many animals from each group to retain and how many to cull (or sell), and how many replacement heifers to purchase, if necessary.

A typical cow spends five years on farm: first year as a calf, second year as a heifer, and the next three years as a milking cow. Assume the herd maintains a fixed structure (structure vector $\boldsymbol{\zeta}_1 = \left[\frac{1}{2}, \frac{2}{5}, 1\right]$), as shown in the second column of Figure B.1. The herd dynamics is characterized by 1 state variable $h_{t,g}$ (the number of milk cows at the beginning of year g of rotation year t) and 1 control variable $\theta_{t,g}$ (the number of milk cows bought in that year). Figure B.1 demonstrates how the herd age cohorts evolve over time. One third of the milking cows are culled every year (cull vector $\boldsymbol{\zeta}_2 = \left[\frac{3}{5}, \frac{1}{15}, \frac{1}{3}\right]$).

The transition equations of herd are

$$\begin{cases} h_{t,g} = h_{t,g-1} + \theta_{t,g} & t = 1, \dots, T, \ g = 2, \dots, G \\ h_{t,1} = h_{t-1,G} + \theta_{t,1} & t = 1, \dots, T, \ g = 1 \end{cases}$$

Initial value $h_{0,G}$ is given.

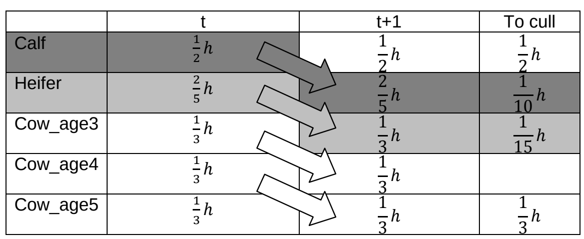

Figure B.1: Herd dynamics at a dairy farm

AppendixB.2. Herd Production

Assume each milk cow consumes a fixed group-specific ration that contains five common components: alfalfa hay, wheat silage, corn grain, soybean meal and protein mix. Also assume that each cow achieves a group-specific weight and produces a fixed amount of milk and waste during each lactation.

Net revenue from herd production:.

$$\pi_t^{herd} = \sum_{g=1}^{G} \Big[p^{milk} \bar{y}_h h_{t,g} - \mathbf{p}^{\mathbf{herd}} \left(\boldsymbol{\zeta}_1 \theta_{t,g} - \boldsymbol{\zeta}_2 h_{t,g} \right)^\top \\ - \left(\mathbf{f}^\top \mathbf{p}^{\mathbf{feed}} + p^{sw} \mathbf{f}^{\mathbf{sw}} + \mathbf{p}^{\mathbf{fixh}} \right) \left(\boldsymbol{\zeta}_1 h_{t,g} \right)^\top - p^M h_{t,g} \Big]$$

- π_t^{herd}, net profit from herd production in rotation-year t [$]
- $\bar{y}_h$, per-cow milk yield [kg/yr]
- $\mathbf{f}$, 5×3 matrix for feed consumption [kg/animal-yr]

- $\mathbf{f^{sw}}$, 3×1 vector for water consumption [m^3/animal-yr]
- p^{milk}, price of milk [$/kg]
- p^{sw}, price of imported surface water [$/$\mathrm{m}^3$]
- p^M, annualized cost of the manure collecting given manure system M [$/cow-yr]
- $\mathbf{p^{herd}}$, 3×1 vector for prices of selling cohorts [$/animal]
- $\mathbf{p^{feed}}$, 5×1 vector for feed price [$/kg]
- $\mathbf{p^{fixh}}$, 3×1 vector for fixed cost [$/animal-yr]

AppendixB.3. Waste Disposal

Assume revenues can be earned from selling dried solid waste but excess liquid waste must be transported off-site at the operator's expense.

Net revenue from waste disposal:.

$$\pi_t^{waste} = \sum_{g=1}^{G} \left[p^{sol} \left(\overline{sol}_{t,g} - L \sum_{k=2g-1}^{2g} sol_{t,k} \right) - \left(p^{base} + p^{dist} r^*_{t,g} \right) \left(\bar{l}_{t,g} - L \sum_{k=2g-1}^{2g} l_{t,k} \right) / \mu^M \right]$$

- π_t^{waste}, net profit from waste management in rotation-year t [$]
- L, the area of cropland on-site [ha]
- $\overline{sol}_{t,g}$, solid waste nitrogen generated in year g of rotation-year t [kgN/yr]
- $sol_{t,k}$, solid waste nitrogen applied on-site during season k of rotation-year t [kgN/ha]
- $\bar{l}_{t,g}$, liquid waste nitrogen generated in year g of rotation-year t [kgN/yr]
- $l_{t,k}$, liquid waste nitrogen apoplied on-site during season k of rotation-year t [kgN/ha]
- $r^*_{t,g}$, the average hauling distance in year g of rotation-year t [km] (refer to the appendix of Baerenklau et al. (2008) for details of the waste disposal cost function)
- p^{base}, the base price for hauling manure off-site [$/ha-cm]
- p^{dist}, the hauling cost per unit distance [$/ha-cm-km]
- p^{sol}, the price received for dried solid waste [$/kgN]
- μ^M, nitrogen concentration of lagoon water given manure system M [kgN/ha-cm]

AppendixC. Crop Model

AppendixC.1. Water, Nitrogen, and Salt Balance Relations

Total nitrogen mass and the salt mass in the mixed irrigation water during season k in rotation-year t at field location j : $n^{w}_{t,k,j}$, $s^{w}_{t,k,j}$

$$
\begin{aligned}
n^{w}_{t,k,j} &= \beta^{I}_{j} \left(sw_{t,k}\, nc^{sw}_{t,k} + gw_{t,k}\, nc^{gw}_{t,k} \right) + prec_{t,k}\, nc^{prec}_{t,k} \\
s^{w}_{t,k,j} &= \beta^{I}_{j} \left(sw_{t,k}\, ec^{sw}_{t,k} + gw_{t,k}\, ec^{gw}_{t,k} + l_{t,k}\, ec^{l}_{t,k}/\mu^{M} \right) + prec_{t,k}\, ec^{prec}_{t,k}
\end{aligned}
$$

$sw_{t,k}$, $gw_{t,k}$, and $prec_{t,k}$ denote the amount of applied surface water [cm-ha/ha], applied groundwater [cm-ha/ha], and precipitation [cm-ha/ha] during season k in rotation-year t. $nc^{a}_{t,k}$ and $ec^{a}_{t,k}$ denote the nitrogen concentration [kg N/cm-ha] and electrical conductivity [dS/m] of water source a, $a \in \{sw, gw, l, prec\}$.

Crop available water [cm-ha/ha], available inorganic nitrogen [kg N/ha], and exposed salinity [dS/m] during season k in rotation-year t at field location j : $w_{t,k,j}$, $n_{t,k,j}$, $s_{t,k,j}$

$$
w_{t,k,j} = \beta^{I}_{j} \left(sw_{t,k} + gw_{t,k} + l_{t,k}/\mu^{M} \right) + prec_{t,k}
$$

$$
\begin{aligned}
n_{t,k,j} = in^{soil}_{t,k,j} + fl_{t,k} + ad_{t,k} + n^{w}_{t,k,j} + \beta^{I}_{j} \left(1-\phi\right) \left(1-\omega\right) l_{t,k} \\
+ \delta_{k} \left(on^{soil}_{t,k,j} + s_{t,k} + \beta^{I}_{j} \omega l_{t,k} \right)
\end{aligned}
$$

$$
\begin{cases}
s_{t,k,j} = \left(s^{soil}_{t,k,j} + s^{soil}_{t,k+1,j} \right)/2, & t = 1,\dots,T;\; k = 1,\dots,K-1;\; j = 1,\dots,J \\
s_{t-1,k,j} = \left(s^{soil}_{t-1,k,j} + s^{soil}_{t,1,j} \right)/2, & t = 2,\dots,T+1;\; j = 1,\dots,J
\end{cases}
$$

$in^{soil}_{t,k,j}$, $on^{soil}_{t,k,j}$, and $s^{soil}_{t,k,j}$ denotes the soil inorganic nitrogen [kg N/ha], soil organic nitrogen [kg N/ha], and soil salinity [dS/m] at the beginning of season k in rotation-year t at field location j. $fl_{t,k}$ is the amount of commercial fertilizer applied [kg N/ha] and $ad_{t,k}$ is the rate of atmospheric nitrogen deposition [kg N/ha] during season k in rotation-year t. ω is the fraction of the organic nitrogen in lagoon water, ϕ is the fraction of applied liquid waste nitrogen that volatilizes during application, and δ_{k} is the seasonal mineralization rate of organic nitrogen.

AppendixC.2. Soil Dynamics

The soil dynamics are characterized by three state variables: $in^{soil}_{t,k,j}$, $on^{soil}_{t,k,j}$, and $s^{soil}_{t,k,j}$

Transition equations:

$$\begin{cases} on^{soil}_{t,k+1,j} = \left(1-\delta_k\right)\left(on^{soil}_{t,k,j} + s_{t,k} + \beta^I_j \omega l_{t,k}\right), & t=1,\ldots,T;\ k=1,\ldots,K-1;\ j=1,\ldots,J \\ on^{soil}_{t,1,j} = \left(1-\delta_k\right)\left(on^{soil}_{t-1,k,j} + s_{t-1,k} + \beta^I_j \omega l_{t-1,k}\right), & t=2,\ldots,T+1;\ j=1,\ldots,J \end{cases}$$

$$\begin{cases} in^{soil}_{t,k+1,j} = \left(1-\lambda_k\right) n_{t,k,j} - nup^R_{t,k,j} - nl^R_{t,k,j}, & t=1,\ldots,T;\ k=1,\ldots,K-1;\ j=1,\ldots,J \\ in^{soil}_{t,1,j} = \left(1-\lambda_k\right) n_{t-1,k,j} - nup^R_{t-1,k,j} - nl^R_{t-1,k,j}, & t=2,\ldots,T+1;\ j=1,\ldots,J \end{cases}$$

$$\begin{cases} s^{soil}_{t,k+1,j} = \dfrac{\nu\, s^{soil}_{t,k,j} + s^w_{t,k,j}}{\nu + w_{t,k,j} - wup^R_{t,k,j}}, & t=1,\ldots,T;\ k=1,\ldots,K-1;\ j=1,\ldots,J \\ s^{soil}_{t,1,j} = \dfrac{\nu\, s^{soil}_{t-1,K,j} + s^w_{t-1,K,j}}{\nu + w_{t-1,K,j} - wup^R_{t-1,K,j}}, & t=2,\ldots,T+1;\ j=1,\ldots,J \end{cases} \tag{C.1}$$

where λ is seasonal denitrification rate, and ν is the amount of water contained in soil at saturation [cm-ha/ha].

Initial values $on^{soil}_{1,1,j}$, $in^{soil}_{1,1,j}$, $s^{soil}_{1,1,j}$, $j = 1, \ldots, J$, are given.

AppendixC.3. Crop Yield and Pollutant Emission

$$\begin{aligned} wup^R_{t,k,j} &= \Psi^c_{wup}\left(w_{t,k,j},\, n_{t,k,j},\, s_{t,k,j}\right) \\ nup^R_{t,k,j} &= \Psi^c_{nup}\left(w_{t,k,j},\, n_{t,k,j},\, s_{t,k,j}\right) \\ ry^R_{t,k,j} &= \Psi^c_{ry}\left(w_{t,k,j},\, n_{t,k,j},\, s_{t,k,j}\right) \\ nl^R_{t,k,j} &= \Psi^c_{nl}\left(w_{t,k,j},\, n_{t,k,j},\, s_{t,k,j}\right) \end{aligned}$$

$ry^R_{t,k,j}$, $wup^R_{t,k,j}$, $nup^R_{t,k,j}$, and $nl^R_{t,k,j}$ are respectively crop relative yield, water uptake [cm-ha/ha], nitrogen uptake [kgN/ha] , and nitrogen leaching [kg N/ha] during season k in rotation-year t at field location j given crop rotation R (let kth crop in rotation R be crop c). Ψ^c are external estimated functions for crop c.

Net revenue from crop production:.

$$\pi^{crop}_t = L\left(\sum_{k=1}^{K}\left[\sum_{j=1}^{J}\left[pr\beta^I_j\, p^R_k\, my^R_k\, ry^R_{t,k,j}\right]\right.\right.$$
$$\left.\left. - p^{sw} sw_{t,k} - p^{gw} gw_{t,k} - p^{rw} rw_{t,k} - p^{fl} fl_{t,k} - p^{fixc}_k\right] - p^I G\right)$$

The net profit from crop production (π_t^{crop}) equals gross returns (crop price times yield) minus both fixed and variable costs. The fixed production costs (p_k^{fixc}) include operating costs such as seed, herbicide, labor, and machinery but not overhead costs. The variable costs include irrigation and fertilizer costs. my_k^R denotes the maximum crop yield in season k given crop rotation R [Mg/ha], and p_k^R the coresponding crop price [\$/Mg]. $sw_{t,k}$, $gw_{t,k}$, $rw_{t,k}$, and $fl_{t,k}$ are respectively applied surface water [ha-cm/ha], applied deep groundwater [ha-cm/ha], recycled shallow groundwater [ha-cm/ha], and applied commercial fertilizer [kgN/ha]. p^{sw} and p^{fl} are the prices of imported surface water [\$/ha-cm] and commercial fertilizer [\$/kgN], while $p^{gw}$and p^{rw} denote the costs of pumping groundwater [\$/ha-cm]. p^I is the annualized cost of an irrigation system given irrigation system I [\$/ha/yr].

Environmental policy cost:.

$$\pi_t^{policy} = L\left(\chi^{sw}\sum_{k=1}^{K}\left[p^{sw}sw_{t,k}\right] + \chi^{fl}\sum_{k=1}^{K}\left[p^{fl}fl_{t,k}\right] + \chi^{fe}\sum_{k=1}^{K}fe_{t,k}^{n} + \chi^{de}\sum_{k=1}^{K}de_{t,k}^{n}\right) + \chi^{sw}\sum_{g=1}^{G}\left[p^{sw}\mathbf{f^{sw}}\left(\boldsymbol{\zeta}_1 h_{t,g}\right)^{\top}\right]$$

Command-and-control policies enter as constraints to the optimization problem. Incentive-based policies, which can include surface water tax (χ^{sw}), commercial fertilizer tax (χ^{fl}), field nitrogen emission charge (χ^{fe}), and downstream nitrogen emission charge (χ^{de}), enter the objective function directly and impose costs on farm production.

AppendixC.4. Irrigation System

Table C.1: Irrigation System Data

Irrigation System Type	Capital Cost [\$/ha]	OM Cost [\$/ha-yr]	Life [year]	Annualized Cost [\$/ha-yr]	CUC	$SD\,[\beta]^*$
½-Mile Furrow	190	6	5	48.68	70	0.3922
¼-Mile Furrow	249	7	5	62.94	75	0.3226
Linear Move	1495	75	12	234.33	90	0.1259

Data source: *University of California Comittee of Consultants*, 1988 (Knapp, 1992)

*The standard deviations for β for are computed for each irrigation system such that $CUC = 1 - \int_0^{\infty} \mid \beta - 1 \mid f\left(\beta\right)\, \mathrm{d}\beta$.

Table C.2: Discretization of the log-normal distribution of water infiltration over the field

Irrigation System Type	β_j^I			$pr\beta_j^I$		
	Under Irrigation	Mean Irrigation	Over Irrigation	Under Irrigation	Mean Irrigation	Over Irrigation
½-Mile Furrow				0.12	0.64	0.23
¼-Mile Furrow	0.3	0.9*	1.7	0.07	0.74	0.18
Linear Move				0.00	0.87	0.13

* The water infiltration coefficient for mean-irrigation is 0.9 rather than 1 due to loss of water along canals, pipelines, etc.

AppendixD. Crop Response Functions

This section summarizes how we formulate and estimate three types of crop response functions: uptake (for water and nitrogen, which are required for the mass balance relationships and transition equations in the whole farm model), relative yield, and field emission. The forms of the uptake and relative yield functions are developed from the traditional Mitscherlich-Baule (MB) form. The field emission function is adapted from the nitrate leaching function in Knapp and Schwabe (2008). Because the estimation methods for the uptake and relative yield functions are quite similar, we only present the latter in this paper. Results show that the methodologies we adopt constitute a reliable approach to estimating these crop response functions with available water (w), available nitrogen (n), and exposed salinity (s) as three input factors.

AppendixD.1. Crop Dataset Generation

We are not aware of any field experiment with data on irrigation, soil nitrogen, nitrogen application rates, soil salinity, crop water uptake, crop nutrient uptake, crop yield, and nitrate leaching. This limited availability of field data is probably due to the high cost of experimentally quantifying the combined effects of multiple input factors on yield and solute leaching. We thus utilize a computer simulation model to generate the data required for estimating the crop response functions.

AppendixD.1.1. Model selection: HYDRUS-1D

Several models have been developed to deal with plant growth, water flow, and solute movement. Pang and Letey (1998) provides a good review. As the authors point out, none of the existing models is adequate to evaluate the integrated effects of water, nitrogen, and salinity on crop yield, either because a model does not use the Richard equation or because a model fails to include one of the effects. They develop a new model, ENVIRO-GRO, to simulate the effects of irrigation depth, irrigation salinity, and nitrogen application on plant yield and nitrogen leaching. The simulated results of ENVIRO-GRO are evaluated against field experiment data, which show good agreements. ENVIRO-GRO might have been an ideal model for the purpose of this study, but the nitrogen component was removed when ENVIRO-GRO was modified to be a user-friendly version.

HYDRUS-1D software package is a modeling environment for analysis of water flow and solute transport in variably saturated porous media (Simunek et al., 2008), which includes most of the underlying mechanisms in ENVIRO-GRO. In fact, HYDRUS-1D offers more functioning and flexibility. It is used worldwide and has been shown to be reliable for modeling water flow and solute transport, especially for processes in soil and in groundwater. A very recent study (Ramos et al., 2011) evaluates HYDRUS-1D using data from a field experiment where corn is irrigated with water of varying nitrogen and salt concentrations. The results show HYDRUS-1D to be a powerful tool for simulating overall salinity and the concentration of nitrogen species in soil.

Ramos et al. (2011) do not consider the active mechanism of root nutrient uptake, which is reasonable given their objective to simulate field conditions in a relatively simple way and give indicative values. Because we are aiming for good quantifications of crop yield and solute leaching, we utilize the compensated root water and nutrient uptake (through both passive and active mechanisms; see Simunek and Hopmans (2009) for details) modules for our study. Originally the Active Solute Uptake module in HYDRUS-1D worked with only one specified solute. However, we need to model the integrated effects of water and two solutes. We collaborated with the HYDRUS-1D developer (Dr. Jiri Simunek, UC Riverside) to modify the program such that the module can handle multiple solutes. Table D.1 summarizes the key specifications of our simulation model in HYDRUS-1D.

Table D.1: HYDRUS-1D specification

Module	**Specification**
Simulated process	water flow, general solute transport, root water uptake, root growth
Soil hydraulic Model	van Genuchten-Mualem
Water Flow Boundary Conditions	Upper: atmospheric BC with surface layer. Lower: free drainage.
Solute Transport	equilibrium model
Solute Transport Boundary Conditions	Upper: concentration flux BC. Lower: zero concentration gradient.
Root Uptake Model	Water Uptake Reduction Model: Feddes. Solute Stress Model: multiplicative model (Threshold Model).
Root Growth Function	50% after 50% growing season

The outputs of HYDRUS-1D include information on water uptake, solute uptake, solute leaching but not on crop yield. External functions are required to relate water and nutrients uptake to crop yield. Following Pang and Letey (1998), relative yield (ry) is specified as a function of relative water uptake and relative nitrogen uptake:

$$ry = \min\left[ry_w, ry_n\right] = \min\left[\frac{w_{up}}{w_{up}^*}, \Phi\left(\frac{n_{up}}{n_{up}^*}\right)\right] \tag{D.1}$$

w_{up} and n_{up} denote the actual amount of water and nitrogen uptake, while w_{up}^* and n_{up}^* denote the potential amount of water and nitrogen uptake (i.e., the maximum water and nitrogen a crop can take up). Φ represents a quadratic relationship (Pang and Letey, 1998, Feng et al., 2005). Using HYDRUS-1D output to calculate relative yields from Equation D.1 gives us full information on crop water uptake, nitrogen uptake, nitrate leaching, and relative yield.

AppendixD.1.2. Model Validation

The best available field experiment data is from a corn trial in Davis, California from 1974 to 1976. The field was treated with 4 different rates of nitrogen fertilizer ($0, 90, 180,$ and $360\,$kgN/ha) and 3 different irrigation regimes ($20, 60,$ and $100\,$cm). See Tanji et al. (1979) and Broadbent and Carlton (1980) for detailed descriptions. We evaluate our approach for modeling root nutrient uptake and calculating relative yield by comparing simulated results with the Davis field data. Soil Salinity is assumed to be $0.01\,$dS/m for all simulations.

Comparisons of field data with simulated results are presented in Figure D.1 and Figure D.2. Linear regression equations are reported along with the coefficients of determination. Figure D.1 displays field measured nitrogen uptake versus the simulated nitrogen uptake from the model of Tanji et al. (1979) and from HYDRUS-1D. The HYDRUS-1D model shows overall better performance than the widely used Tanji model. The slope coefficient is closer to one and the intercept term is closer to zero and quite small relative to the range of nitrogen uptake. The null hypotheses that these coefficients are respectively equal to one and zero can neither be rejected at 95% confidence level. Figure D.2 compares the simulated relative yield from HYDRUS-1D to field data. Although the fit is not as good as that for simulated nitrogen uptake, it is still quite good given the complexities and uncertainties associated with the whole plant-water-nitrogen-salinity system. The R^2 value compares favorably against previously reported measures in the literature that are in the range of 0.6-0.8. In summary, the results demonstrate the ability of HYDRUS-1D to accurately model root uptake and validate our approach to simulating relative yield.

AppendixD.1.3. Dataset Generation

For each crop, we simulate combinations of at least 5 levels of available water ($[0.25, 0.5, 1, 1.5, 2] \times w_{up}^*$), 5 levels of available inorganic nitrogen ($[0.25, 0.5, 1, 1.5, 2] \times n_{up}^*$), and 6 levels of soil salinity ($[0, 0.2, 0.4, 0.6, 0.8, 1] \times \overline{EC}$), which produces at least 150 scenarios. $\overline{EC}$ [dS/m] is a critical value of soil salinity at which crop yield decreases to zero. These combinations are selected to cover most, if not all, of a farm operator's anticipated farming practices under current conditions as well as NMPs or other hypothetical nutrient policies. Section D.3 takes

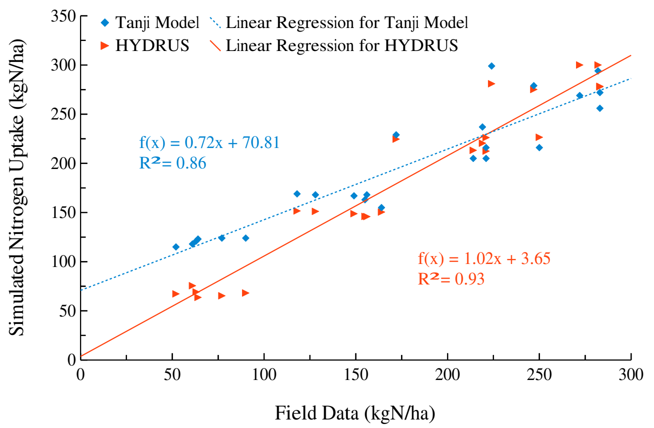

Figure D.1: Field measured nitrogen uptake vs. simulated nitrogen uptake from Tanji et al. (1979) model & from HYDRUS-1D

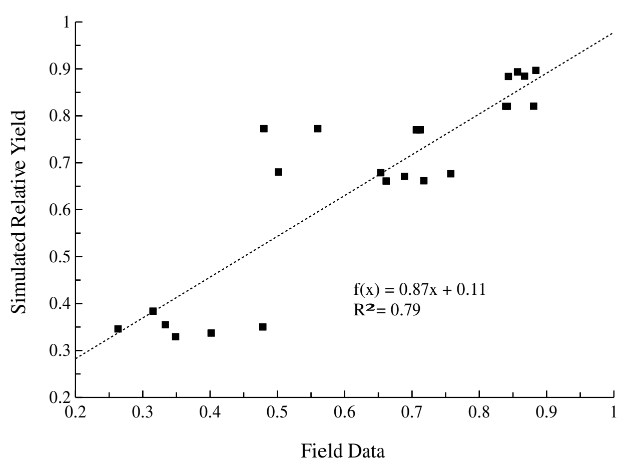

Figure D.2: Field measured relative yield vs. simulated relative yield from HYDRUS-1D corn as an example to illustrate the logic behind function modifications and the approximation procedure.

AppendixD.2. Function Specification and Estimation

AppendixD.2.1. Relative Yield Function

Choice of Functional Form. Various forms have been proposed for crop yield functions. Griffin et al. (1987) gives a very good review on twenty traditional and popular functional forms. They also discuss guidelines for form selection, one of which pertains to application-specific characteristics. Because the resulting functions in this study are to be used in economic optimization procedures, continuous differentiability is a desirable property. Llewelyn and Featherstone (1997) compares five functional forms using corn yield data from the CERES-Maize simulator for western Kansas. Corn yield is estimated against nitrogen and irrigation water. Their results favor the Mitscherlich-Baule (MB) form over all other specifications. Also, they show that the costs of incorrectly using the MB form is relatively low. Shenker et al. (2003) measures the yield response of sweet corn to the combined effects of nitrogen fertilization and water salinity over a wide range of nitrogen and salinity levels. Two functional forms are evaluated based on the measured data: Liebig–Sprengel (i.e., linear von Liebig) and MB. The results suggest that either functional form can successfully predict water needs, nitrogen needs, and yield. Liebig–Sprengel is a minimization function derived from von Liebig's "law of the minimum". It results in a stepwise response curve which is not differentiable. Therefore, MB is chosen as the base functional form for relative yield in this study.

The traditional MB function is usually expressed as Equation D.2, where a represents a plateau of the production level Y and b_i are parameters for the input factor X_i. This function exhibits continuously positive marginal productivities of input factors and allows for factor substitution. Following this form, relative yield as a function of three inputs can be written as Equation D.3. Either absolute yield y or relative yield ry (the ratio of the actual yield y over the maximum yield y^*) can be the dependent variable, with a equal to y^* for using absolute yield and a equal to one for relative yield. $b_w, b_n,$ and b_s are respectively parameters for water, nitrogen, and salinity.

$$Y = a \prod_i \left(1 - \exp\left(-b_i^1\left(X_i + b_i^0\right)\right)\right) \tag{D.2}$$

$$ry \equiv \frac{y}{y^*} = \left(1 - \exp\left(-b_w^1\left(w - b_w^0\right)\right)\right)\left(1 - \exp\left(-b_n^1\left(n - b_n^0\right)\right)\right) \left(1 - \exp\left(-b_s^1\left(s - b_s^0\right)\right)\right) \tag{D.3}$$

Function Modification and Estimation. Unfortunately, Equation D.3 fails to fit the data well. We find that, given a salinity level and a nitrogen level, the simulated yields have bell-shaped distributions over the full range of water level (Figure D.3). Therefore, we introduce a parameterized variant of the logistic probability density function, called the water coefficient φ, into the water component of the function. The logistic distribution is preferred over the normal distribution because of its heavier tails. Also shown in Figure D.3, the bell shapes vary for different salinity levels. Equation D.3 is thus modified so that salinity enters the function through its influence on the water and nitrogen parameters rather than directly as a separate multiplicative term.

The relative yield function is defined as:

$$ry = \left(1 - \exp\left(-b_w^1\left(\varphi w - b_w^0\right)\right)\right)\left(1 - \exp\left(-b_n^1\left(n - b_n^0\right)\right)\right) \tag{D.4}$$

where

$$\varphi = \frac{\exp\left(d_1 w + d_0\right)}{4\left(1 + \exp\left(d_1 w + d_0\right)\right)^2} + d_2$$

With this specification, each parameter in $\boldsymbol{\Upsilon} \equiv \left\{b_w^1, b_w^0, b_n^1, b_n^0, d_0, d_1, d_2\right\}$ is effectively a function of salinity. This approach reduces the computation requirement by breaking down the problem into two subproblems.

Subproblem 1 Estimate Equation D.5 once for each value of $s = 0, 6, 12, 18, 24, 30$ dS/m, using the appropriate subset of simulated data points.

$$ry = g^{ry}(w, n, \boldsymbol{\Upsilon}) \tag{D.5}$$

See Equation D.4 for the explicit form of g^{ry}. These estimations produce the surfaces shown in Figure D.3. The figures show excellent agreement between simulated data (points) and fitted data (surfaces) at each salinity level.

Subproblem 2 Estimate each parameter $\Upsilon_i \in \boldsymbol{\Upsilon}$ as a polynomial function of salinity, as shown in Equation D.6. Figure D.4 depicts the regression curves and Table D.2 reports the estimated functions. Again, agreement between the data (point estimates) and functions is generally very good.

$$\Upsilon_i = f_i^{ry}(s) \tag{D.6}$$

Substitute fitted Equation D.6 into Equation D.4 to get relative yield function G^{ry}. This approach is verified by the good agreement between simulated data and the fitted relative yield from G^{ry} (Figure D.5).

$$ry = g^{ry}(w, n,\ f^{ry}(s)) = G^{ry}(w, n, s) \tag{D.7}$$

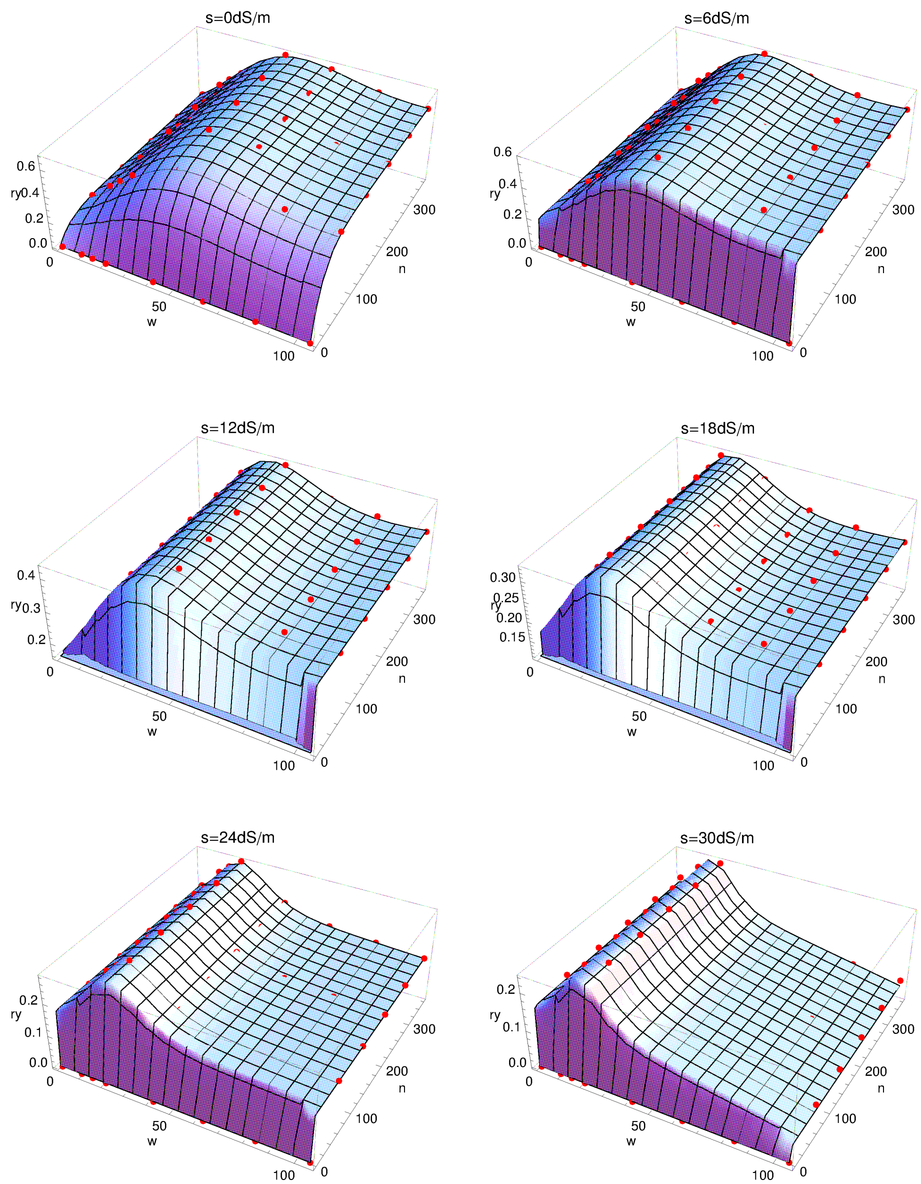

Figure D.3: Relative yield vs. available water and available nitrogen when soil salinity is 0, 6, 12, 18, 24, and 30 dS/m. Points: simulated data from section D.1. Surfaces: fitted functions.

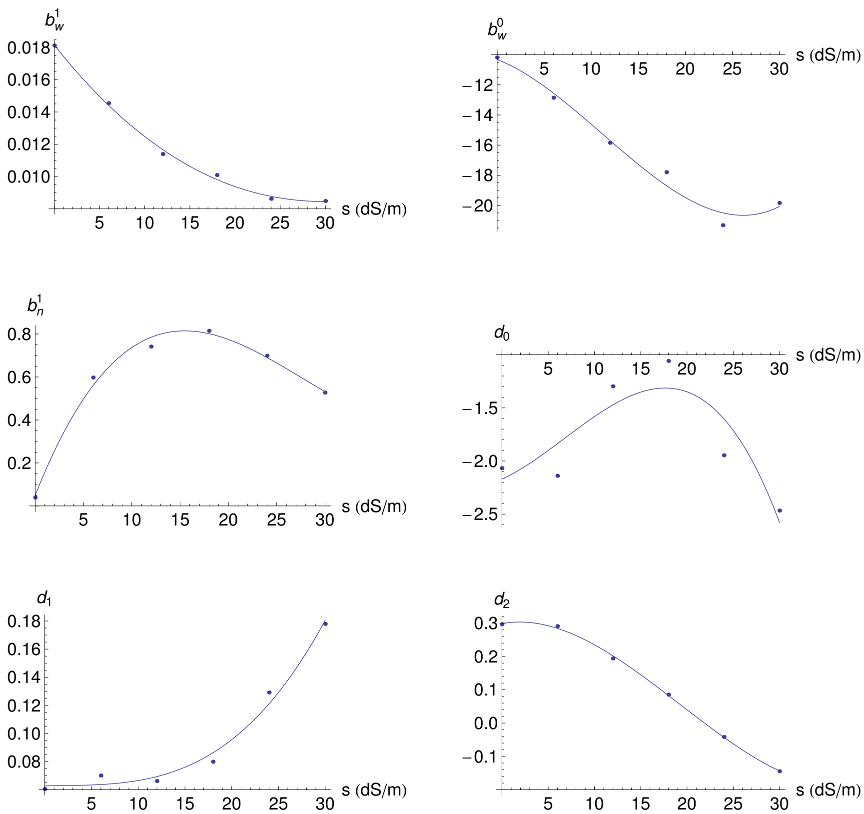

Figure D.4: Polynomial regression of water and nitrogen parameters in the relative yield function

Table D.2: Polynomial regression of water and nitrogen parameters in the relative yield function

Υ_i			$f_i^{ry}(s)$		
	1	s	s^2	s^3	R^2
b_w^1	0.0181	-0.0007	-	-	0.9973
b_w^0	-10.3161	-0.2408	-0.0270	0.0008	0.9815
b_n^1	0.0495	0.1136	-0.0051	0.0001	0.9908
d_0	0.0627	0.0001	-	-	0.9846
d_1	-2.1708	0.0398	0.0038	-0.0002	0.7736
d_2	0.2987	0.0050	-0.0014	-	0.9993

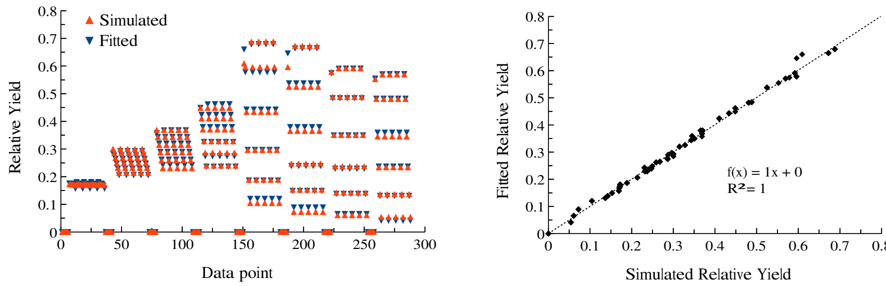

Figure D.5: Relative yield function: simulated data vs. fitted data

AppendixD.2.2. Field Emission Function

We test several forms for the field emission (nitrate leaching) function. The function adapted from Knapp and Schwabe (2008) outperforms the quadratic, cubic, and square root functions, mainly because of its convex-concave behavior and guarantee of a plateau maximum. In Knapp and Schwabe (2008), nitrogen leaching is specified as a function of initial soil nitrogen, applied nitrogen, and infiltrated water. Equation D.8 is a simplified version, where nl is the amount of field emissions [kgN/ha] and n is the total available nitrogen [kgN/ha]. Similar to the specification of the relative yield function, $\boldsymbol{\vartheta} \equiv \left\{\vartheta_w^1, \vartheta_w^0, \vartheta_n\right\}$ are parameters that depend on salinity levels.

$$nl = \frac{\vartheta_n \cdot n}{1 + \exp\left(-\vartheta_w^1 \left(w - \vartheta_w^0\right)\right)} \tag{D.8}$$

$$\vartheta_i = f_i^{nl}\left(s\right) \tag{D.9}$$

We adopt the same procedure to estimate the field emission function. Equation D.8 is estimated for $s = 0, 6, 12, 18, 24, 30$ dS/m (Figure D.6). Equation D.9 is then estimated $\forall \vartheta_i \in \boldsymbol{\vartheta}$ and substituted into Equation D.8 to generate the field emission function $G^{nl}(w, n, s)$ (Table D.3, Figure D.7). Figure D.8 shows that the estimated field emission function fits the simulated data well.

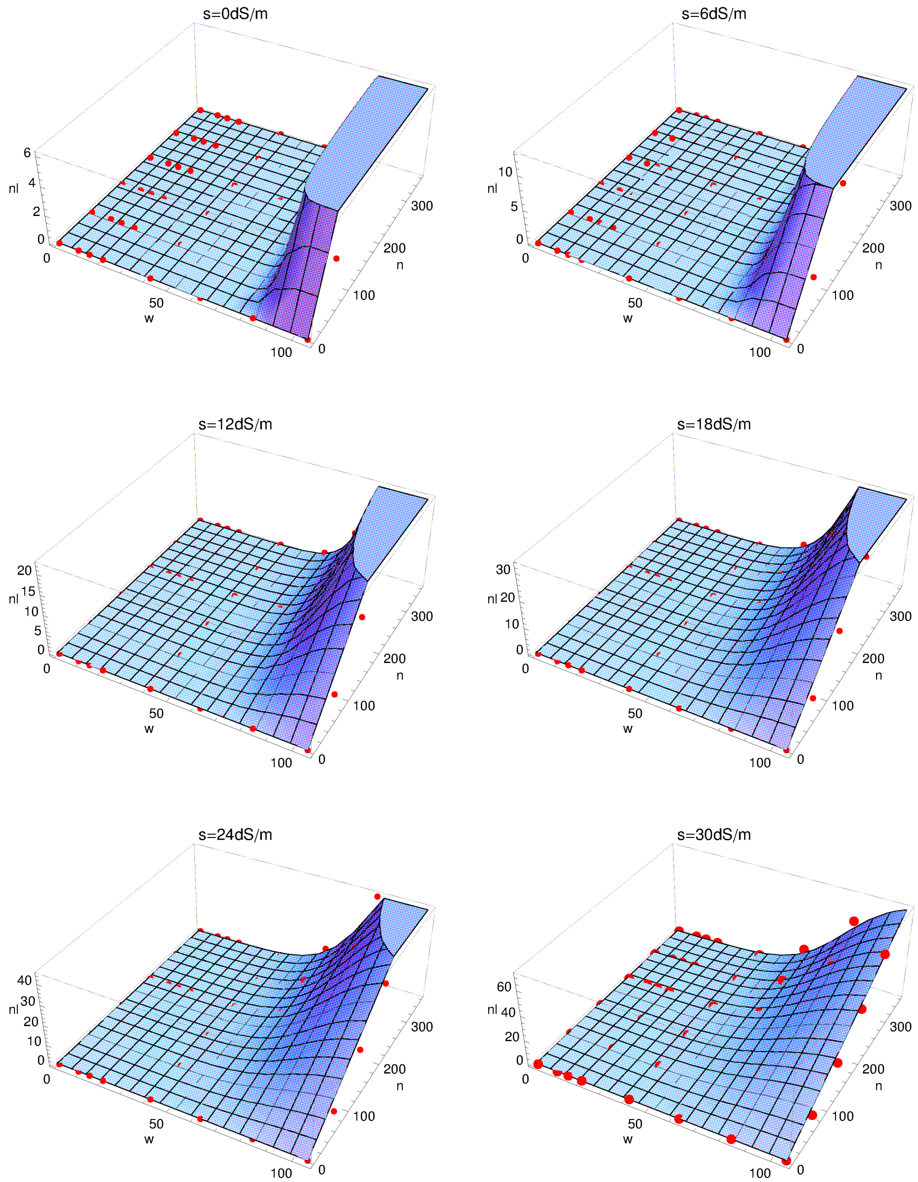

Figure D.6: Field emission vs. available water and available nitrogen when soil salinity is $0, 6, 12, 18, 24,$and $30\,\mathrm{dS/m}$. Points: simulated data from section D.1. Surfaces: fitted functions.

Table D.3: Polynomial regression of water and nitrogen parameters in the field emission function

ϑ_i	$f_i^{nl}(s)$			
	1	s	s^2	R^2
ϑ_w^1	0.4147	-0.0232	0.0004	0.9067
ϑ_w^0	88.1392	0.0046	-0.0087	0.9734
ϑ_n	0.0888	0.0066	-0.0001	0.9878

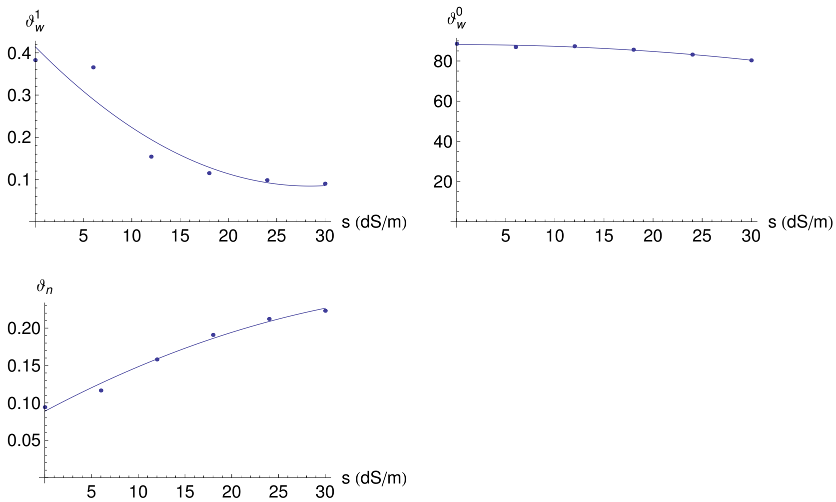

Figure D.7: Polynomial regression of water and nitrogen parameters in the field emission function

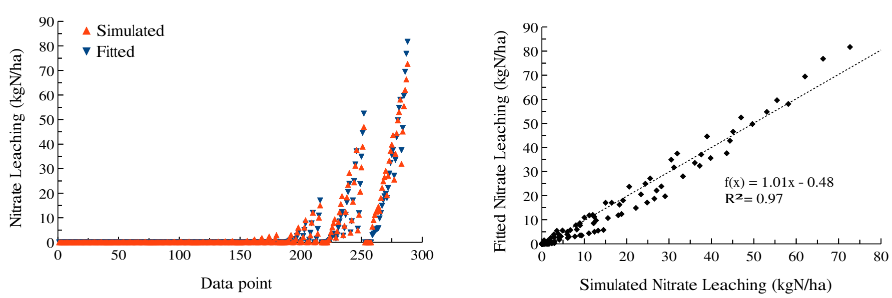

Figure D.8: Field emission function: fitted data vs. simulated data

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