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Polices for Controlling Groundwater Pollution from Concentrated Animal Feeding Operations

A Dissertation submitted in partial satisfaction
of the requirements for the degree of

Doctor of Philosophy

in

Environmental Sciences

by

Jingjing Wang

June 2012

Dissertation Committee:

Professor Kenneth Baerenklau, Chairperson
Professor Kurt Schwabe
Professor Keith Knapp

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The Dissertation of Jingjing Wang is approved:

Committee Chairperson

University of California, Riverside

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To my parents.

ABSTRACT OF THE DISSERTATION

Polices for Controlling Groundwater Pollution from Concentrated Animal Feeding Operations

by

Jingjing Wang

Doctor of Philosophy, Graduate Program in Environmental Sciences
University of California, Riverside, June 2012
Professor Kenneth Baerenklau, Chairperson

Animal waste from animal feeding operations (AFOs) is a significant contributor to nitrate contamination of groundwater. Some animal waste also contains heavy metals and salts that may build up in cropland and underlying aquifers. This thesis focuses on pollution reduction from the largest AFOs, in particular, Concentrated Animal Feeding Operations (CAFOs), which present the greatest potential risk among all AFOs to environmental quality and public health. To find cost-effective policies for controlling pollution at the field level and at the farm level, a dynamic environmental-economic modeling framework for representative CAFOs is developed. The framework incorporates four models (i.e., animal model, crop model, hydrologic model, and economic model) that includes various components such as herd management, manure handling system, crop rotation, water sources, irrigation system, waste disposal options, and pollutant emissions. The operator maximizes discounted total farm profit over multiple periods subject to environmental regulations. Decision rules from the dynamic optimization problem demonstrate best management practices for CAFOs to improve their economic and environmental perfor-

mance. Results from policy simulations suggest that direct quantity restrictions of emission or incentive-based emission policies are much more cost-effective than the standard approach of limiting the amount of animal waste that may be applied to fields; reason being, policies targeting intermediate pollution and final pollution create incentives for the operator to examine the effects of other management practices to reduce pollution in addition to controlling the polluting inputs. Incentive-based emission policies are shown to have advantages over quantity restrictions over multiple years when seasonal or annual emissions fluctuate either due to inherent operation practices or the accumulation of precursors to the pollution. My approach demonstrates 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. It also suggests that ecosystem services can play an important role in pollution control and thus deserve more attention when designing policies.

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Part I

Introduction

Chapter 1

Background

1.1 Animal Feeding Operations

1.1.1 AFOs and CAFOs

The growing world population, together with globally converging diets, has fuelled 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, Oenema et al., 2005]. During the same period per capita meat consumption in the developing countries rose by 150 percent and that of dairy products by 60 percent [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 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]. Higher farm incomes due to economies of scale will sustain the trend toward larger and more concentrated animal feeding operations, both in developed and rapidly growing economies.

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 leads to less land available for animal waste disposal, which is the primary method of disposal. In addition, animal waste (especially dairy and swine manure) is costly to move relative to its nutrient value. Therefore, the common practice among operators continues to be over-application of animal waste on land close to the facility. Excess nutrients from this over-application tend to be transported off the farm where they can produce adverse environmental and health effects.

In the United States, as of 2003, there are an estimated 1.3 million farms with livestock, of which 238,000 are considered animal feeding operations [USEPA, 2003]. An animal feeding operation (AFO) is defined as:

> A lot or facility (other than an aquatic animal production facility) where the following conditions are met: (1) Animals have been, are, or will be stabled or confined and fed or maintained for a total of 45 days or more in any 12-month period, and (2) crops, vegetation, forage growth, or post-harvest residues are not sustained in the normal growing season over any portion of the lot or facility [USEPA, 2003, Page 7188].

An important category of AFOs is concentrated animal feeding operations (CAFOs) - the largest of the animal operations and the one that poses the greatest risk to environmental quality and public health, which are subject to strict environmental regulations. CAFOs are categorized into a three-tier structure::

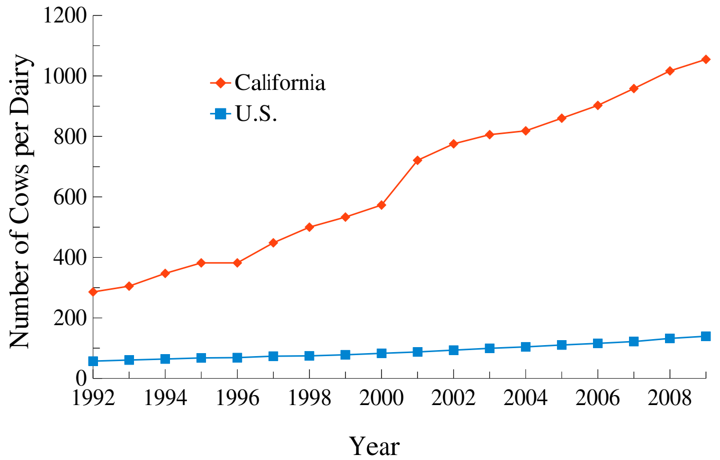

Figure 1.1: Average number of cows per dairy in the U.S. and California

> [A]ll large operations are CAFOs, medium operations are CAFOs if they meet specified risk-of-discharge criteria, and small operations are CAFOs only if they are so designated by EPA or the State NPDES permitting authority [USEPA, 2003, Page 7189].

The size thresholds for defining Large, Medium, and Small CAFOs in each sector are specified. Take the dairy sector as an example: a facility confining 700 or more mature dairy cattle is a Large CAFO, a facility confining 200 to 699 mature dairy cattle is a Medium CAFO and a facility confining less than 200 mature dairy cattle is a Small CAFO [USEPA, 2003, Table 4.1].

1.1.2 Environmental and Public Health Impacts

In the United States, AFOs annually produce more than 500 million tons of animal waste[USEPA, 2003]. The composition of waste at a particular operation depends on the animal species, size, maturity, and animal feed. The main constituents are the same: nutrients (particularly nitrogen and phosphorus), organic matter, solids, pathogens, and volatile compounds [USEPA, 2003]. When improperly managed, animal waste can pose substantial risks to public health and ecological systems, as discussed below.

1.1.2.1 Water Pollution

Nutrients Animal waste contains significant quantities of nitrogen and phosphorus. Both nitrogen and phosphorus have fertilizer value for crop growth, but either can also produce adverse environmental impacts when it is transported in excess quantities to the environment. Nutrient pollution is a leading cause of water quality impairment in lakes, rivers, and estuaries [USEPA, 2000]. Nitrogen and phosphorus accelerate algae production in receiving aquatic ecosystems and can result in algal blooms, and is associated with a variety of problems including clogged pipelines, fish kills, and reduced recreational opportunities [USEPA, 2000].

Besides harming surface water, nitrate-nitrogen reaching groundwater is a potential threat to public health. Two medical conditions have been linked to excessive concentration of nitrate in drinking water: methaemoglobinaemia ('blue-baby syndrome') in infants, and stomach cancer in adults [Addiscott, 1996, Bower, 1978]. As of 2000, about 43.5 million people (15 percent of U.S. population) rely on domestic withdraws as their source of drinking water, primarily from groundwater [Hutson et al., 2004]. These people are at greater risk of nitrate poisoning than those relying on public water sources, because the quality and safety of water from domestic wells are not regulated by the Safe Drinking Water Act or, in many cases, by state laws [USEPA, 2003, DeSimone et al., 2009]. In 2009, the National Water-Quality Assessment (NAWQA) Program of the U.S. Geological Survey published a report that presents a national assessment of water quality in private domestic wells based on samples from about 2,100 wells located in 48 states from 1991 to 2004 [DeSimone et al., 2009]. One of the major reported findings is that nitrate is the most common contaminant derived from anthropogenic sources that is found at concentrations greater than human-health benchmarks. Nitrate occurs naturally in ground water, but elevated concentrations usually originate from human activities such as fertilizer application, animal production, and septic systems [Nolan and Hitt, 2003]. According to the NAWQA report, concentrations of nitrate are greater than the EPA maximum contaminant level (MCL) of nitrogen in 4.4 percent of wells throughout the country. The highest concentrations of nitrate occur most frequently in aquifers underneath agricultural regions, such as Central Valley basin-fill aquifers in California and the Basin and Range aquifers in the Southwest [DeSimone et al., 2009].

Salinity Salinity is a measure of the amount of dissolved salts or ions in the water. Two typical indexes are used to measure salinity: Total Dissolved Solids (TDS) reported as mg/L (milligrams per liter) and Electrical Conductivity (EC) reported as dS/m (deci-Siemens per meter). In this thesis, the EC designation is used.

The salinity of animal waste is directly related to the presence of salts such as sodium, potassium, calcium, magnesium, chloride, sulfate, bicarbonate, carbonate, and nitrate, which are from undigested feed that passes unabsorbed through animals [USEPA, 2003]. When animal waste is applied to land, salts build up in the soil and ultimately accumulate in receiving ecosystems.

Soil salinization is a common problem in areas with low rainfall. High salinity in the soil can deteriorate soil structure, reduce permeability, restrict plant roots from withdrawing water due to osmotic forces and thus reduce crop yields; when combined with irrigation and poor drainage, it can lead to permanent soil fertility loss [USEPA, 2003]. Soil salinity can be maintained at acceptable levels for crop growth by applying excess water to leach the salts out of soil. However, this management strategy results in excess deep percolation flows which can create their own set of problems, including the salinization of the receiving water bodies such as aquifers or streams and rivers. Two strategies have been proposed to deal with such issues: improvement of irrigation management so that excess water is not applied over that needed for evapotranspiration and leaching, and reuse of drainage waters for irrigation of appropriate salt-tolerant crops [Qadir and Oster, 2004].

Salts in water also can have adverse impacts on public health and drinking water supplies. Even at low levels, salts can increase blood pressure in salt-sensitive individuals, increasing their risk of stroke and heart attacks [USEPA, 2003]; or if delivered for public use, even slightly saline water can lead to increased water treatment costs, water loss, and pipe maintenance [Helperin et al., 2001].

Other Pollutants Animal waste contains other pollutants that can impair estuaries, lakes, streams and rivers. Organic compounds reaching surface water reduce dissolved oxygen, which can decrease biodiversity and kill fish. Dissolved solids can lead to surface water degradation. More than 150 pathogens found in livestock manure pose risks to humans, including the six human pathogens that account for more than 90 percent of food and waterborne diseases in humans [USEPA, 2003]. Trace elements, antibiotics, pesticides and hormones are also potential contaminants contained in animal waste.

1.1.2.2 Air Pollution and Greenhouse Gases Emissions

Numerous airborne contaminants (e.g., gases, dust, and microbes) are produced by or emitted from animal production facilities and their waste disposal practices [Jacobson et al., 1999]. Animal housing and manure handling systems generate a variety of gases[1], but only three of them have been studied in detail: hydrogen sulfide, ammonia, and methane. Hydrogen sulfide poses the largest safety risk in confined spaces, ammonia can result in ecological damage to the environment, and methane contributes to global warming. Other gases, such as volatile fatty acids and oxides of nitrogen, are currently being studied in greater detail because of their contribution to odor or their potential impact on global warming [Jacobson et al., 1999]. Recent studies focus on nitrous oxide (N_2O) offset in crop production for GHG emission reductions in industry sectors by changing fertilizer management practices [Millar et al., 2010]. As of 2002, AFOs contributed approximately 3 percent of all U.S. nitrous oxide (similar for methane) emissions [USEPA, 2003].

Little information is available on the environmental impact of dust or microbe emissions. Most research has focused on implications for the indoor air quality affecting both animals and humans [Jacobson et al., 1999].

1.1.3 California Dairies

California has been the nation's leading dairy state since 1993 when it surpassed Wisconsin in milk production. Its 1.84 million dairy cows produced 41.63 billion pounds of milk in 2011, generating 21.1 percent of the national supply [CDFA, 2012]. Thirty five

[1] Livestock Buildings can generate up to 168 different compounds, accordnig to O'neill and Phillips [1992].

counties contributed to the state's marketed milk production, of which the top five counties (Tulare, Merced, Kings, Stanislaus, and Kern) accounted for 71.1 percent of the state's total milk production in 2011 [CDFA, 2012]. However, these cows generate over 30 million tons of manure each year, so proper management of dairy waste on California's dairy farms is one of the state's most pressing environmental issues [USEPA, 2011].

In the 1990s, California's dairy industry experienced significant growth and concentration. In 1993, California's 4000 dairies produced 2.7 billion gallons of milk; in 1998, 2700 dairies produced 20 percent more milk. During the same time period, the average number of cows per dairy increased from 367 to 624 [USEPA, 2011]. The trend towards large farms continues in the new century. The average size of a dairy herd in California was 656 cows in 2002 and in 2011 it was 1101 cows [CDFA, 2003, 2012]. In the Kern County, the average number of cows in a dairy operation is up to 3069 [CDFA, 2012]. Increasing concentration and intensifying production of dairies lead to the concentration of dairy waste in specific geographic areas, creating a potential threat to California environmental and public health.

1.1.4 California Groundwater Contamination

California uses more groundwater than does any other state in the nation, extracting a daily average of 14.5 billion gallons of groundwater [Helperin et al., 2001]. In 2000, approximately 50 percent of California's population depended on groundwater for its drinking water supplies [Helperin et al., 2001]. In an average year, groundwater meets about 25 to 40 percent of California's urban and agricultural water demands; in drought years, this percentage can increase to two thirds [Helperin et al., 2001]. Groundwater clearly is an important natural resource for California and therefore it is imperative to develop appropriate policies to safeguard this vital resource from contamination.

According to the California Department of Water Resources, three-fourths of the state's impaired groundwater is contaminated by nitrates, salinity, and pesticides [Helperin et al., 2001]. Nitrates are the most common type of nutrient contamination in California, having caused the closure of more public wells than any other contaminant [Helperin et al., 2001]. As NRDC states in their 2001 report [Helperin et al., 2001], "by 1994, nitrate levels more than double the acceptable limit resulted in the closures of more than 800 wells in Southern California and 130 in the San Joaquin Valley alone." In some other areas, though not as significant, nitrates are also among the primary pollutants in public water supply wells. Salinity is also a severe problem in the agricultural regions of California. Between 400,000 and 700,000 acres of arable land in California are estimated to be lost by 2010 as a result of increasing water and soil salinity [Helperin et al., 2001]. In the San Joaquin Valley of California, about 39 percent of the arable and productive land is saline [Knapp and Baerenklau, 2006].

Most of California's dairy farms are located in the Central Valley and the mismanagement of dairy waste has led to groundwater degradation [Hurley et al., 2007]. As described in the Waste Discharge Requirements General Order for Existing Milk Cow Dairies (General Order) published by the Central Valley Regional Water Quality Control Board in 2007,

> Groundwater monitoring shows that many dairies in the Region have impacted groundwater quality. A study of five dairies in a high-risk groundwater area in the Region found that groundwater beneath dairies that were thought to have good waste management and land application practices had elevated levels of salts and nitrates beneath the production and land application areas.

> The Central Valley Water Board requested monitoring at 80 dairies with poor waste management practices in the Tulare Lake Basin. This monitoring has also shown groundwater pollution under many of the dairies, including where groundwater is as deep as 120 feet and in areas underlain by fine-grained sediments [CRWQCB, 2007, pp.6-7].

1.2 Existing Policy Regime

The major Federal environmental law currently affecting animal feeding operations is the Clean Water Act (CWA). CWA establishes a comprehensive program for protecting the nation's waters. Among its core provisions, it prohibits the discharge of pollutants from a point source to waters of the United States except as authorized through a National Pollutant Discharge Elimination System (NPDES) permit. The Act also requires EPA to establish national technology-based effluent limitations guidelines and standards (ELGs) for different categories of sources.

Agriculture has been recognized primarily as a nonpoint pollution source and exempted from NPDES requirements. Nevertheless, parts of an animal operation, such as the farmstead (animal production facilities not including the adjacent lands), are easily identified and more similar to point sources. Therefore, CWA has historically defined the term ''point source'' to include CAFOs of more than 1000 animal units (section 502, CWA). In the mid 1970s, EPA established ELGs and permitting regulations for CAFOs under the NPDES program [USEPA, 2003]. Similar to traditional point pollution sources in other industries, CAFOs were required to install acceptable technologies at the site to improve farmstead structures and control runoff. The exception was the irrigation of wastewater to crop fields. At that time, the regulations presumed that manure removed from the production area was handled appropriately through land application.

Despite more than three decades of regulation of AFOs, reports of discharge and runoff of animal waste from these operations persist [USEPA, 2003]. Although this is in part due to inadequate compliance with and enforcement of existing regulations, the changes that have occurred in the animal production industries contribute more to the persisting waste discharge. The continued trend toward fewer but larger operations via more intensive production methods, coupled with shrinking acreage for hay and pastures, is concentrating more animal waste within smaller geographic units. A high correlation has been found between areas with impaired surface and/or ground water due to nutrient enrichment and areas where dense livestock exist [USEPA, 2003].

In response to these concerns, USDA and EPA announced in 1999 the Unified National Strategy for Animal Feeding Operations. The Strategy establishes the goal that "all AFO owners and operators should develop and implement technically sound, economically feasible, and site specific comprehensive nutrient management plans (CNMPs) to minimize impacts on water quality and public health" [USDA and USEPA, 1999, pp.5]. A comprehensive suite of voluntary programs and regulatory programs are geared to ensure that AFOs establish appropriate CNMPs for properly managing animal manure, including on-farm application and off-farm disposal, if any. As specified in the Strategy, voluntary programs (e.g. locally led conservation, environmental education, and financial assistance, and technical assistance) address the vast majority of AFOs, while the regulatory program focuses on high risk operations [USDA and USEPA, 1999].

To approach the goals of the Strategy, EPA published a new rule for CAFOs in 2003. This rule can be seen as a part of the regulatory program proposed by the Strategy. It expands the number of CAFOs required to seek NPDES permit coverage up to 15,500 operations, including 11,000 large CAFOs [USEPA, 2003]. One important change is that the revised ELGs require large CAFOs to prepare and implement site-specific nutrient management plans (NMPs) for animal waste applied to land. The guidelines for NMPs include land application rates, setbacks, and other land application Best Management Practices (BMPs), which are required to be based on the most limiting nutrient for applying fertilizer to cropland [USEPA, 2003]. NMPs would be nitrogen-based in areas where soil phosphorus is low and phosphorus-based where soil phosphorus is high [USEPA, 2003]. The nutrient standard can limit animal waste application rates on most land. Without better methods for animal waste disposal other than land application, NMPs will increase competition for land capable of absorbing animal waste and create additional costs for farm operators.

EPA finalized the rule in 2008 in response to the order issued by the U.S. Court of Appeals for the Second Circuit in Waterkeeper Alliance et al. v. EPA. There are two changes relative to the 2003 CAFO regulations. First, only those CAFOs that discharge or propose to discharge are required to apply for permits; second, CAFOs are required to submit the NMPs along with their NPDES permits applications, which will then be reviewed by both permitting authorities and the public [USEPA, 2008]. Nevertheless, the fundamental restrictions in NMPs remain the same for CAFOs as in the 2003 rule. For a thorough review of federal and state regulations for water pollution from land application of animal waste, refer to Centner [2012].

Atmospheric pollutants are regulated under the Clean Air Act (CAA), but CAA currently does not recognize AFOs for regulatory purposes. Despite the slow progress on regulatory change in practice, air pollution from AFOs is receiving increasing attention in the academic literature, especially when cross-media effects of some pollutants are taken into account [Aillery et al., 2005, Baerenklau et al., 2008, Sneeringer, 2009]

1.3 Animal Waste Management

1.3.1 Animal Waste Management Strategies

1.3.1.1 Land Disposal

Nitrogen, phosphate, and potash contents in animal waste represent a potential substitute for commercial fertilizers on field crops. However, the concentration of CAFOs in specific regions implies that more facilities are competing for the same amount of land. Generally, off-site transportation of animal waste over long distances is costly relative to its nutrient value. Also, cropland operators may be reluctant to receive animal waste from nearby CAFOs because of the odor and uncertainties (e.g. the amount of nutrient contents, weeds, and pathogens) associated with the application of the "organic" fertilizer. When combined, these factors lead to over-application of animal waste on land nearest to the facility.

In a study of the economics of dairy manure use as fertilizer in Central Texas, Adhikari et al. [2005] conclude that, at the current costs for loading, hauling, and spreading, dairy manure cannot be economically transported from surplus to deficit areas within the study area. Their results suggest that off-site land application of dairy manure has the potential to be profitably substituted for chemical fertilizers, if appropriate subsidies are paid, either by the government or by dairy operators, or if the moisture content of the manure is reduced.

Paudel et al. [2009] use a GIS-based model to determine the least cost dairy manure application pattern for Louisiana's major dairy production area where environmental quality and crop nutrient requirements are treated as constraints. According to their analysis, the characteristics of dairy manure limit the distribution areas or distances between the farms and the land over which the manure can be economically spread. Longer distances between dairies and farmland favor the use of commercial fertilizers on farmland.

1.3.1.2 Wetland Treatment

Wetlands provide a chemical and biological environment suitable for improving water quality. Constructed wetlands have been used to improve the quality of river water, stormwater, coal mine drainage, and municipal sewage [Schaafsma et al., 1999]. The technology of constructed wetland has been applied and evaluated at dairy farms of relatively small size (less than 200 cows). For large CAFOs, at least to the author's knowledge, the potential of wetlands as an alternative treatment for animal waste has neither been theoretically studied nor empirically investigated.

According to Geary and Moore [1999], a treatment wetland, constructed to reduce organic matter and nutrients in dairy parlor waters, does not appear to be suitable as a treatment option for significantly reducing nutrients in dairy wastewater at a herd size of 110. Schaafsma et al. [1999] discuss a wetland system for treating wastewater from a dairy farm (170 cows) in Maryland. The analysis of the water samples show that flow through the wetland system resulted in significant reductions in concentrations of all analytes except nitrate/nitrite. Relative to initial concentrations, total nitrogen is reduced 98 percent but nitrate/nitrite increase by 82 percent. One of the possible solutions they propose is recirculation of wastewater through the wetland cells to promote denitrification and uptake of nutrients by plants. The constructed wetland has also been tested on very small animal farms. Dunne et al. [2005] find that wetland performance is seasonally variable at a 42-cow organic dairy unit in Ireland with large open space and this variability is primarily controlled by hydrological inputs. MacPhee et al. [2009] adopt a diffused air aeration system for a constructed wetland receiving dairy wastewater. They concludes that the benefits of wetland aeration are not great enough to warrant its widespread adoption for small-scale agricultural systems.

1.3.1.3 Anaerobic Digester System

The waste from dairy and swine operations could support the operation of anaerobic digester systems [USEPA, 2003]. Digestion primarily transforms the content of animal waste to a different form, resulting in small reductions in overall volume [Simpkins, 2005]. Benefits to operators using anaerobic digesters include the cost savings from electricity generation, control of methane emission, significant odor reduction, and improved marketability of the digester solids [USEPA, 2003]. However, anaerobic digesters would not necessarily reduce the nutrients in animal waste. Digesters take available nitrogen and convert part of it into ammonium in the liquid byproduct, which is more readily available to plants [USDA, 2007a]. Most of the phosphorus removed from the effluent is concentrated in the digested solids [USEPA, 2003]. Through anaerobic digester systems, animal waste can be made more valuable and less likely to become an odor problem, but remains subject to land application requirements.

Morse et al. [1996] conduct an anaerobic digester survey of California dairy producers to investigate the failure of many previously installed methane recovery systems. Identified problems include poor design, collection of manure in a wet form, and incomplete cooperation from electricity companies.

Hurley et al. [2007] focus on clustering of independent dairy operations for generation of bio-renewable energy. The financial feasibility of regional anaerobic digesters, which capture methane from dairy manure produced from cows in multiple farms, is empirically analyzed. Their results emphasize the importance of trucking cost as an impediment to centralized digesters under the common flush manure handling system in the Central Valley. However, as noted in their paper, "As the EPA and regional water control boards tightens the constraints on how dairy producers dispose of their manure, producers over time may find that these regulations push them to change their current flush systems to scrape systems to more efficiently manage the disposal of the manure [Hurley et al., 2007, pp.vi]." Alternative manure management systems are considered in this thesis as a potential option that farm operators might take to comply with environmental regulations.

1.3.2 Dairy Manure Management in California

Morse et al. [1996] conduct 139 written and 45 oral surveys to identify practices for the collection, storage, and use of manure on California dairy farms in Tulare, Fresno, and Madera counties, where mean milking herd size ranges from 381 to 910 cows. As presented in their paper,

> Liquid wash or flush waters were stored in ponds on 95.9% of the dairies. Settling ponds or basins (39.1%) or mechanical solid separators (14.2%) were used to reduce the solid loading rate in storage ponds. Manure solids were collected

> by tractors (solid system) or from settling ponds or basins (liquid system). Few producers (4.1%) identified composting as a component of manure handling. Liquid manures were used for year-round irrigation (62.2%), spread as slurry (9.5%), sold or transported off the farm (12.2%), or seasonally irrigated (62.2%). Solid manure was spread on farm land (78.4%), used for bedding (27.0%), sold off the farm (58.1%), removed from the farm (6.8%), or composted (5.4%).

In summary, around 90 percent of the manure from California dairies, liquid and solid, is applied to land on the farm.

Operators in the Central Valley apparently have chosen flush dairy systems rather than scrape systems, due to the relatively low cost of water in comparison to labor [Hurley et al., 2007]. The characteristics of the dairy manure in California, its bulk and relatively low primary nitrogen, phosphate, and potash levels, generally make it infeasible for most dairies to participate in a centralized digester or constructed wetland treatment. Land application remains the most common and most desirable method of utilizing manure at California's large dairies.

Chapter 2

Literature Review

2.1 Environmental-Economic Analysis for CAFOs

Since EPA published the final rule for CAFOs in 2003, 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: the cost of transporting and spreading manure to a specific amount of receiving land, and the 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 associated with the savings from replacing commercial fertilizers [Ribaudo et al., 2003]. Kaplan et al. [2004]

utilize a sector model to evaluate regional adjustments in production 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 a 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], arguably the most complete and accurate study ever performed, 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.

Another set of studies examines alternative policies for animal waste management at the regional level, triggered by the manure policy intervention in the European Union. For example, the Dutch Quota System has received substantial attention in literature [Vukina and Wossink, 2000, Wossink and Gardebroek, 2006, Helming and Reinhard, 2009]. Straeten et al. [2011] use the Flemish policy case to compare a concentration permit trading system with a regular emission permit trading system. The manure spreading model under the concentration permit is formulated in a framework similar to the classical warehouse location problem in operation research. The results show better emission spreading at higher costs under the concentration permit trading system.

2.2 Policy Instruments for Groundwater Protection

The only policy instrument for controlling pollution from CAFOs that has been discussed in the literature is the NMPs proposed by EPA. However, the problem of over-application of animal waste to land is a classic agricultural nonpoint source pollution problem. Therefore, a review of the policies, which have been proposed for the general problem of nitrogen pollution and salinity from agricultural production, can shed some light on the appropriate policies for controlling groundwater pollution from CAFOs.

2.2.1 Nitrates in Groundwater

Four types of regulatory targets have been considered to control nitrate pollution of groundwater in the literature: nitrogen input to the land surface [Berntsen et al., 2003], direct effluent from the agricultural field/ nitrate leaching from the root zone [Wu et al., 1994, Helfand and House, 1995, Larson et al., 1996], nitrogen surplus from the agricultural system [Berntsen et al., 2003, Cuttle and Jarvis, 2005], and ambient concentration in the groundwater aquifer [Almasri, 2007, Peña-Haro et al., 2009]. For each target, two broad categories of control mechanisms can be implemented: "command and control (CAC)" (standards and liability rules) or incentive-based instruments (e.g., taxes, subsidies, quotas).

The NMPs established by EPA is an example of the traditional CAC approach. NMPs require that the land application rates of animal waste must be consistent with agronomic rates of nutrient uptake by crops. Since NMPs set quantity restrictions upon the "last-stage" input to the pollution process, it does provide an incentive for farm operators to reduce herd numbers, change feed, or choose different crops to grow on farm. However, some other management practices (e.g. irrigation methods, wastewater recycling) are overlooked because emissions are not directly regulated, implying potential cost ineffectiveness of NMPs to reduce groundwater pollution [Baerenklau et al., 2008].

Wu et al. [1994] develop a dynamic model to simulate farmers' choices (of crops and irrigation systems) and the resulting levels of farm income and nitrogen runoff/percolation. Four policies to reduce water pollution are simulated: (a) a tax on nitrogen runoff and percolation; (b) a nitrogen use tax; (c) restrictions on irrigation water use; and (d) cost sharing for adopting modern irrigation technologies. An effluent tax on nitrogen runoff and percolation is shown to be effective in reducing nitrate pollution, while a tax on nitrogen use is shown to be the least effective policy. The efficacy of the other two policies depends on soil type. Similarly to [Baerenklau et al., 2008], these results also suggest the NMP requirement for CAFOs might not be the most cost-effective policy to reduce pollution.

Helfand and House [1995] show that several uniform regulatory instruments can achieve a pollution target at relatively low social cost. Their case study site is the Salinas Valley of California, where fertilizer applied to lettuce production leads to buildup of nitrate in groundwater. Five types of policy instruments are analyzed as different methods of achieving a 20 percent pollution reduction: (a) separate input taxes for each soil type and each input (a solution which achieves the social optimum); (b) tax both inputs to production, with taxes uniform across soil types; (c) require a uniform percentage rollback in levels of input use; (d) tax either water or nitrogen uniformly across soil types; and (e) restrict either water or nitrogen use uniformly across soil types. The socially optimal solution to (a) is presented only as a benchmark of comparison, which is likely to be infeasible in practice under heterogeneous conditions. Three of the second-best policies (the water tax, the policy of identical input taxes for both soil types, and the uniform reduction in all input use) are nearly as efficient as the use of individual input taxes. Under the uniform nitrogen tax and the nitrogen use restriction, the welfare loss is the largest relative to the overall revenues from lettuce production. Larson et al. [1996] later identify water to be the best single input to tax in this case study.

Fleming and Adams [1997] consider an ambient tax based on groundwater nitrate concentrations at observation well sites. In the empirical study of irrigated agriculture in a county of Oregon, the change in farm profitability under a tax that incorporates spatial differences in physical parameters is compared with that under a uniform tax. The results indicate that the gains from accounting for spatial variance in physical parameters are quite modest.

Goetz et al. [2006] bring attention to the distorting effect of intensive margin policies on the extensive margin. They empirically show that combining a nitrogen input tax with land-use taxes is about 18 percent more cost efficient than a nitrogen input tax alone and 58 percent more efficient than offsite abatement in the form of groundwater treatment.

2.2.2 Salts in Groundwater

For aquifers beneath irrigated agriculture, three concepts are closely related: irrigation drainage, soil salinity, and groundwater salinity. Unrestricted irrigation drainage has led to environmental degradation in many areas worldwide, especially in lands such as California's San Joaquin Valley. Furthermore, if drainage is improperly controlled, salts may accumulate in the soil and impact crop productivity.

Soil salinity and drainage generation have a substantially developed literature in economics [Dinar et al., 1993, Knapp, 1999]. Dinar et al. [1993] evaluated five policy instruments that have been proposed to address drainage and salinity problems in central California: (1) surface water tax, (2) drainage tax, (3) surface water quota, (4) drainage quota, and (5) irrigation technology cost sharing. They show that in general, direct policies targeting drainage will achieve drainage goals more cost effectively than indirect policies targeting water use that contributes to drainage.

Scant attention has been paid by economists to groundwater salinization. Knapp and Baerenklau [2006] provide a brief review. In their paper, a long-term economic-hydrologic model of agriculturally induced groundwater salinization is implemented to determine efficient management in the presence of both pumping costs and salt externalities. Pricing instruments (one for salt emissions and one for groundwater extractions) are set to ensure the efficient outcome.

2.3 Summary

Policy provisions for nitrate and salt pollution in the previous studies take a variety of forms: quotas applied to polluting outputs or contributing inputs; taxes levied directly on polluting outputs or indirectly on contributing inputs; and public cost sharing for improved input or pollution management technologies. Although these studies are based on the pollution from crop agriculture, they can provide some insights into the control of pollution from CAFOs. A tax on nitrogen use and a control on nitrogen application have been shown to be the least effective policy [Wu et al., 1994, Helfand and House, 1995], suggesting the nutrient restrictions on CAFOs proposed by EPA are probably not the appropriate policy to reduce groundwater pollution. Instead, an effluent tax on nitrogen runoff or a water tax might be more cost-effective.

The fact that regulations on water or irrigation systems can affect nutrient pollution and salinization is not surprising, because water is the media of transport for both salts and nitrates. For crop agriculture relying upon commercial fertilizers, nutrient pollution and salinization are treated as disparate issues. However, when animal manure containing both nitrogen and salts is applied to land as organic fertilizer, the two problems should be addressed simultaneously. As discussed above, except for Baerenklau et al. [2008], few studies undertake field-level and farm-level analyses of controlling nutrient emissions from CAFOs. This may be due to the fact that there is very limited information on how crop yields and leaching rates respond to varying application rates of water, nitrogen, and salts.

Part II

Integrated Farm Level Model

The preceding two chapters present the challenges of controlling groundwater pollution from CAFOs and of estimating the economic impacts on CAFOs that must comply with environmental regulations. In the following four chapters, I construct an integrated farm level model to address these challenges.

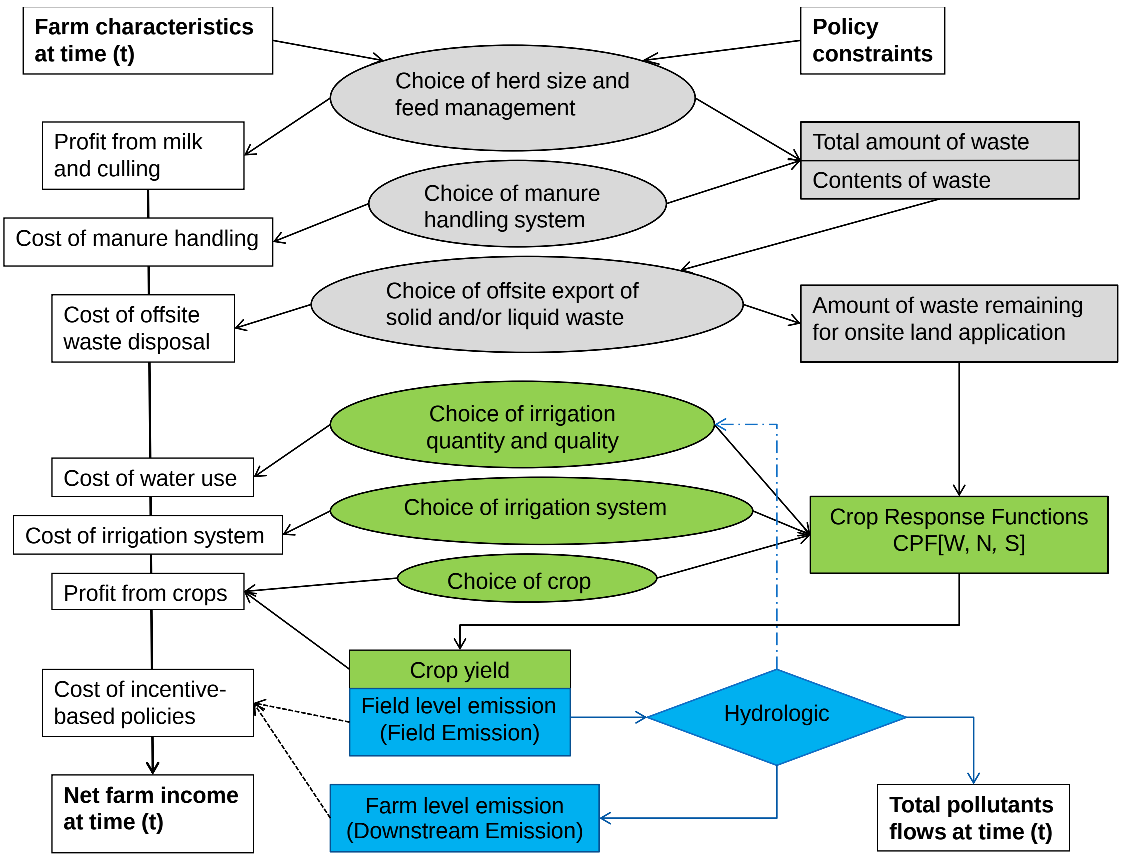

Figure 2.1: Key elements of the integrated farm level model (adapted from Baerenklau et al. [2008])

The 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. The three main building blocks that make up the full model are animal, crop, and hydrological models. The animal model and the crop model have corresponding economic submodels, which together constitute the whole farm economic model. The hydrological model simulates the pollutant emission both at the field level and at the farm level. When incentive-based policies are imposed, the cost of these policies also enters the economic model.

Chapter 3

Animal Model

3.1 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.

3.1.1 Herd Dynamics

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 my model does not include the formal transition equations for each age cohort. Instead, I 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.

The operator works in discrete time and manages the herd. Each year the operator decides how many animals to retain and how many to cull (or sell), and how many replacement animals 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 ($calf : heifer : cow = \frac{1}{2} : \frac{2}{5} : 1$), as shown in the second column of Figure 3.1. The herd dynamics are characterized by 1 state variable $h_{t,g}$ (the number of milk cows at the beginning of year g of time period t)[1] and 1 control variable $\theta_{t,g}$ (the number of milk cows bought in that year). Figure 3.1 demonstrates how the herd age cohorts evolve over time. The numbers of calves, heifers, and cows that are culled every year are respectively $\left(\frac{1}{2}+\frac{1}{10}\right)h_{t,g}$, $\frac{1}{15}h_{t,g}$, and $\frac{1}{3}h_{t,g}$. Define a structure vector $\boldsymbol{\zeta}_1 = \left[\frac{1}{2}, \frac{2}{5}, 1\right]$ and a cull vector $\boldsymbol{\zeta}_2 = \left[\frac{3}{5}, \frac{1}{15}, \frac{1}{3}\right]$.

The transition equations of herd are

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

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

[1] I adopt three time indices in the model: time period t, year g, and season k. As discussed in Chapter 4, crop rotation is over multiple years. Therefore, I use t to cover a full period of crop rotation. For example, the alfalfa-corn rotation usually takes 6 years. In this case there are 6 years and 12 seasons in one time period. See Appendix A for the notations.

	t	t+1	To cull
Calf	$\frac{1}{2}h$	$\frac{1}{2}h$	$\frac{1}{2}h$
Heifer	$\frac{2}{5}h$	$\frac{2}{5}h$	$\frac{1}{10}h$
Cow_age3	$\frac{1}{3}h$	$\frac{1}{3}h$	$\frac{1}{15}h$
Cow_age4	$\frac{1}{3}h$	$\frac{1}{3}h$	
Cow_age5	$\frac{1}{3}h$	$\frac{1}{3}h$	$\frac{1}{3}h$

Figure 3.1: Herd dynamics at a dairy farm

3.1.2 Herd Production

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.

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.

3.1.3 Economic Submodel for Herd Production

The net profit from herd production is equal to the revenue from milk and meat production less the total production cost.

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

- π_t^{herd}, net profit from herd production in time period t [$/time-period]
- $\bar{y}_h$, per-cow milk yield [kg/yr]
- $h_{t,g}$, the number of milking cows at the beginning of year g of time period t
- $\theta_{t,g}$, the number of milking cows bought at the beginning of year g of time period t
- $\boldsymbol{\zeta}_1$, structure vector
- $\boldsymbol{\zeta}_2$, cull vector
- $\mathbf{f}$, 5×3 matrix for feed consumption [kg/animal/yr]
- $\mathbf{f^{sw}_M}$, 3×1 vector for water consumption given manure handling system M [m^3/animal/yr]
- p^{milk}, price of milk [$/kg]
- p^{sw}, price of imported surface water [$/m^3]
- p^M, annualized cost of 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]

3.2 Animal Waste

3.2.1 Waste Generation and Management

Animal operations produce waste slurry. Some waste solids are separated and sold off-site as fertilizer. For the flows of waste nitrogen, refer to Figure A1 in Baerenklau et al. [2008]. The remaining liquid waste needs to be disposed, usually in one of the three ways: land application, wetland treatment, or 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 I assume 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 willingness to accept manure (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 more labor and capital intensive but use much less water per cow compared to the flush-lagoon system and thus produces a smaller volume of waste. The two also differ 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 presumably it can be uniformly applied over the field[2]. 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., 2007]. 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 241.77m^3/cow/yr and that of a scrape-tank system is 131.24m^3/cow/yr [Bray et al., 2011].

3.2.2 Economic Submodel for Waste Management

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

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

- π_t^{waste}, net profit from waste management in time period t [\$/time-period]
- L, the area of cropland on-site [ha]
- $\overline{sol}_{t,g}$, solid waste nitrogen generated in year g of time period t [kgN/yr]
- $sol_{t,k}$, solid waste nitrogen applied on-site during season k of time period t [kgN/ha]

[2] In practice, farmers solve an optimization problem to determine the spreading pattern on site since it costs more to transport waste to further parts of the field. Therefore, tractor spreading is not perfectly uniform. In such a case, there are two sources of spatial heterogeneity: irrigation and animal waste spreading. New parameters and control variables can be introduced in the future to take into account this "dual spatial heterogeneity".

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

Chapter 4

Crop Model

4.1 Economic Submodel for Crop Production

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.

$$\pi_t^{crop} = L\left(\sum_{k=1}^{K}\left[\sum_{j=1}^{J}\left[pr\beta_j^I\, p_k^R\, my_k^R\, ry_{t,k,j}^R\right] - p^{sw} sw_{t,k} - p^{gw} gw_{t,k} - p^{rw} rw_{t,k} - p^{fl} fl_{t,k} - p_k^{fixc}\right] - p^I G\right)$$

$pr\beta_j^I$ accounts for the non-uniformity of irrigation systems, which are discussed in Section 4.2. 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]. $ry_{t,k,j}^R$ denotes the relative crop yield during season k in time period t at field location j given crop rotation R. The relative yield functions are estimated from simulated crop dataset, which are demonstrated in Section 4.4 and 4.5. $sw_{t,k}$, $gw_{t,k}$, $rw_{t,k}$, 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].

4.2 Irrigation

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 $E\left[\beta\right] = 1$ and standard deviation $SD\left[\beta\right]$. Data on common irrigation systems is readily available from previous studies. $SD\left[\beta\right]$ can be calculated from the Christiansen uniformity coefficient of each system. See Table 4.1 for irrigation system data and the calculation of $SD\left[\beta\right]$. All costs are expressed in 2005 dollars.

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_j, $j \in \{1, 2, 3\}$. I 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. The corresponding probabilities are $pr\beta_j^I$, $j \in$

Table 4.1: Irrigation System Data

Irrigation System Type	Capital Cost [$/ha]	OM Cost [$/ha-yr]	Life [year]	Annualized Cost [$/ha-yr]	CUC	$SD\,[\beta]^{*}$
1/2-Mile Furrow	327	10	5	83.73	70	0.3922
1/4-Mile Furrow	428	12	5	108.26	75	0.3226
Linear Move	2571	129	12	403.05	90	0.1259

Data source: *University of California Committee of Consultants*, 1988 [Knapp, 1992]
*The standard deviations for β for are computed for each irrigation system such that $CUC = 1 - \int_0^\infty \mid \beta - 1 \mid f(\beta)\ \mathrm{d}\beta$.

$\{1, 2, 3\}$. See Table 4.2 for discretized intervals over the field.

4.3 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 response functions in the literature has taken into account the effects of interactions and feedback mech-

Table 4.2: 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
1/2-Mile Furrow				0.13	0.64	0.23
1/4-Mile Furrow	0.3	0.9*	1.7	0.08	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.

anisms 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 as well as root water and nutrient uptake to generate simulated crop datasets, from which both yield and emission functions are developed. I 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.

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 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 analyses 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 I model the crop system following the first approach.

4.4 Crop Dataset Generation

I am not aware of any field experiment with a full set of data on irrigation, soil nitrogen, nitrogen application rates, soil salinity, crop water uptake, crop nutrient uptake, crop yield, and nitrate leaching, which is required for estimating the crop response functions. 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. I thus utilize a computer simulation model to generate the required data, and use available data to validate the approach.

4.4.1 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. Given the objective of quantifying of crop yield and solute leaching, I utilize the compensated root water and nutrient uptake modules for my study (through both passive and active mechanisms; see Simunek and Hopmans [2009] for details). Originally the Active Solute Uptake module in HYDRUS-1D worked with only one specified solute. However, I need to model the integrated effects of water and two solutes. I collaborated with the HYDRUS-1D developer (Dr. Jiri Simunek, UC Riverside) to modify the program such that the module can handle multiple solutes. For modification manual, see Appendix B. Table 4.3 summarizes the key specifications of my simulation model in HYDRUS-1D.

Table 4.3: 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
Root Growth Function	50% after 50% growing season

The outputs of HYDRUS-1D include information on water uptake, solute uptake, and 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:

$$ry = \min\left[ry_w, ry_n\right] = \min\left[\frac{w_{up}}{w^*_{up}}, \Phi\left(\frac{n_{up}}{n^*_{up}}\right)\right] \tag{4.1}$$

w_{up} and n_{up} denote the actual amount of water and nitrogen uptake, while w^*_{up} and n^*_{up} denote the the maximum potential uptake of water and nitrogen. Φ represents a quadratic relationship [Pang and Letey, 1998, Feng et al., 2005]. Using HYDRUS-1D output to calculate relative yields from Equation 4.1 gives us full information on crop water uptake, nitrogen uptake, nitrate leaching, and relative yield.

4.4.2 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 $360\,\mathrm{kgN/ha})$ and 3 different irrigation regimes $(20, 60,$ and $100\,\mathrm{cm})$. See Tanji et al. [1979] and Broadbent and Carlton [1980] for detailed descriptions. I evaluate my approach for modeling root nutrient uptake and calculating relative yield by comparing simulated results with the Davis field data. Because the Davis trial did not consider variable salinity conditions, soil Salinity is assumed to be $0.01\,\mathrm{dS/m}$ for all simulations.

Comparisons of field data with simulated results are presented in Figure 4.1 and Figure 4.2. Linear regression equations are reported along with the coefficients of determination. Figure 4.1 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 not be rejected at 95% confidence level. Figure 4.2 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 the previously reported value in the literature ($R^2 = 0.84$ in Pang and Letey [1998]). In summary, the results demonstrate the ability of HYDRUS-1D to accurately model root uptake and validate my approach to simulating relative yield.

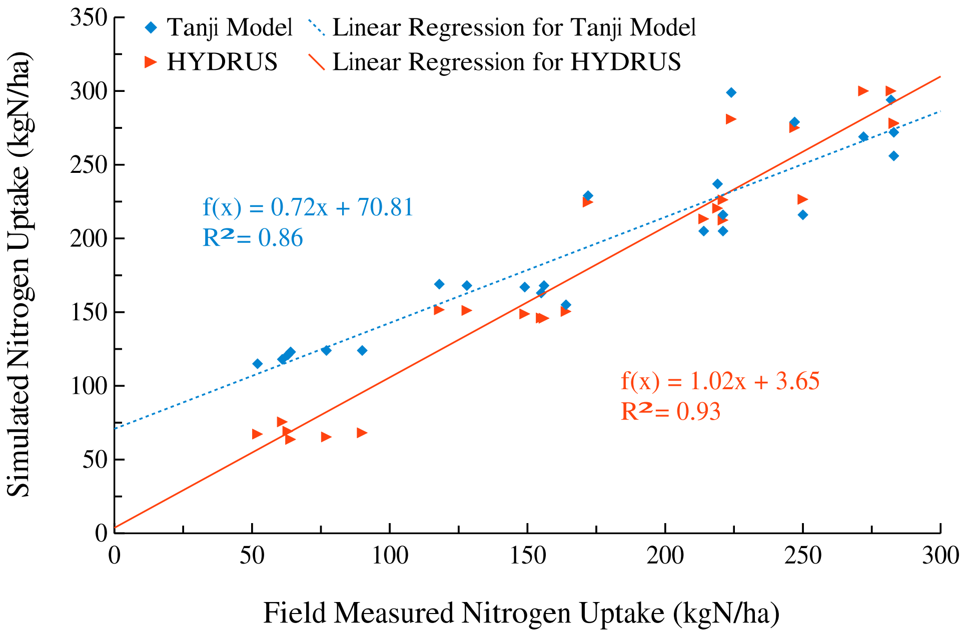

Figure 4.1: Field measured nitrogen uptake vs. simulated nitrogen uptake from Tanji et al. (1979) model & from HYDRUS-1D

4.4.3 Model Input

Required Data Input There is considerable input data needed for each selected crop: corn, cotton, safflower, sunflower[1], small grains (wheat, Bartali oat, Swan oat, Longhorn oat). See Appendix B for how I collect the raw data and calculate the intermediate data, which are briefly summarized here.

- Raw data
  - crop: growing days, crop evapotranspiration coefficient (K_c), salt tolerance, maximum root depth, maximum nitrogen uptake, maximum water uptake
  - climate: average daily temperature, precipitation, reference evapotranspiration (ET_0)

[1]Estimated response functions for corn, cotton, and small grains are reported in Appendix C. Estimation of the functions for safflower and sunflower encountered currently unresolved numerical problems in the second stage regression and thus are not included in the crop rotation choice. Therefore, the policy results presented in this thesis provide a lower bound on the cost savings of switching from NMPs to other policies.

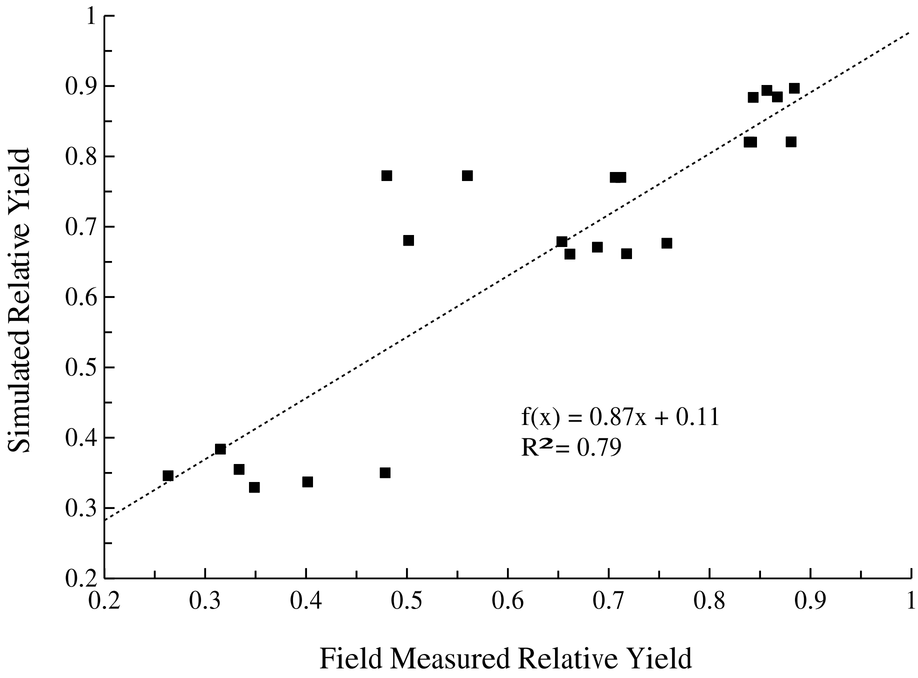

Figure 4.2: Field measured relative yield vs. simulated relative yield from HYDRUS-1D

- soil: soil type, bulk density, residual soil water content, saturated soil water content, saturated hydraulic conductivity, etc.

- Intermediate data (calculated from raw data)
  - water uptake curve (daily potential water uptake): daily crop evapotranspiration $ET_c = K_c \times ET_0$
  - nitrogen uptake curve (daily potential nitrogen uptake)

Linear Approximation of Nitrogen Uptake Curve There is little information on the nitrogen uptake curves of crops other than small grain forages that are discussed in Crohn et al. [2009]. My idea is to use the available data on small grain forages to test approximating different shapes of nitrogen uptake curves via a linear curve; if the approximation is good, I can then use a linear relationship to calculate the daily potential nitrogen uptake based on the maximum nitrogen uptake of a crop and its growing degree-days (GDD) over the season.

Figure 4.3 depicts the nitrogen uptake curves for 8 small grain forages commonly grown on California dairies. These curves are based on the logistic function developed by Crohn et al. [2009], where the nitrogen content of a crop is a function of GDD. In practice, the nitrogen uptake of a crop might be higer than the crop's nitrogen content, since some nitrogen can be lost to the atmosphere. Here I assume that the nitrogen uptake and the nitrogen content of a crop are identical. The curves fall into two categories: sigmoid and exponential. However, as Crohn et al. [2009] point out, the exponential shapes of two ryegrass crops are most likely due to forage quality and harvest schedule constraint.

In order to investigate whether the shape of a curve significantly affects actual nitrogen uptake and relative yield, the simulation results of Swan oat (top curve in Figure 4.3), Longhorn oat (bottom curve in Figure 4.3), and Bartali Italian ryegrass (middle curve in Figure 4.3) are compared to the simulation results of a linearized version of the crops (the same crop but with a linear nitrogen uptake curve). All specifications in the HYDRUS-1D model are exactly same for the four crops except for the daily potential nitrogen uptakes, which depend on the nitrogen uptake curve.

Figure 4.4 displays the simulation results. The "outliers" are scenarios where crops receive sufficient water with low salinity but little nitrogen. The t-statistics suggest that the relative yield of the hypothetical crop is statistically equal to relative yield of Longhorn oat and Bartali Italian ryegrass, and is statistically equal to 98.68% of the relative yield of Swan oat. I conclude that the effect of the shapes of nitrogen uptake curve is not statistically significant thereby a linear relationship can be used to approximate nitrogen uptake curves for my purposes.

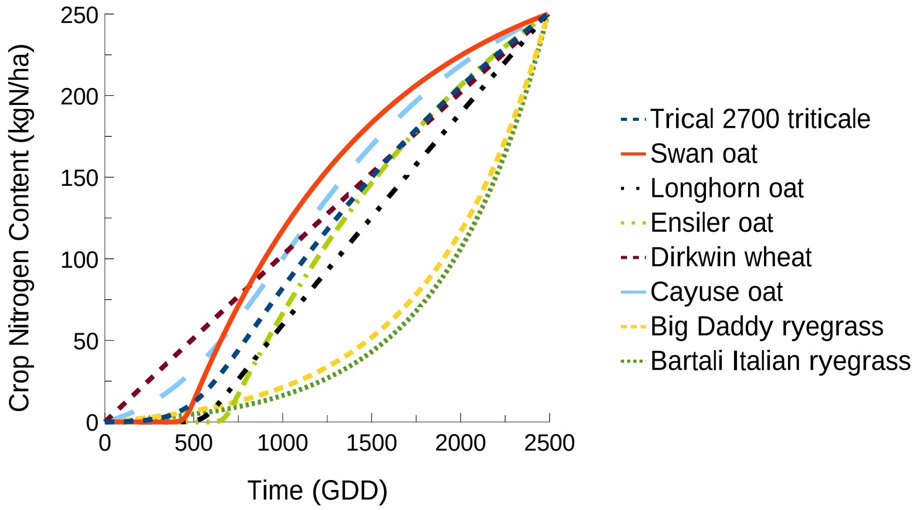

Figure 4.3: Nitrogen uptake curves for eight small grain forages commonly grown in California

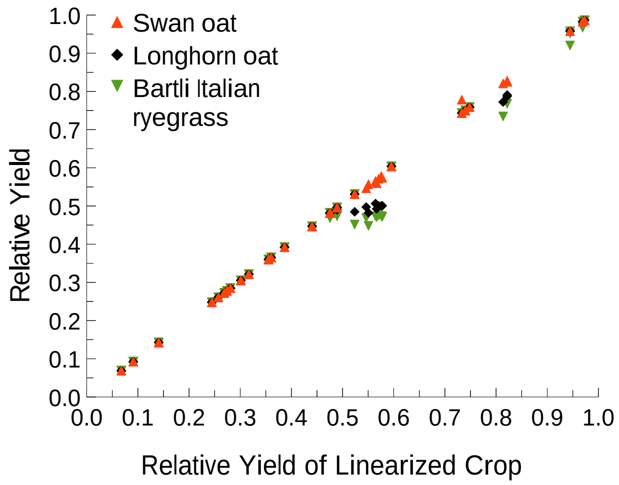

Figure 4.4: Relative yield of the linearized crop vs. relative yield of Swan oat, Longhorn oat, and Bartali Italian ryegrass

4.4.4 Dataset Generation

For each crop, I 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 for each crop. These levels are selected in order to cover most, if not all, of a farm operator's possible farming practices even under the hypothetical regulatory scenarios to be posed later. The highest level of water is double maximum water uptake, taking into account that the operator might apply excess water to flush salts out of the root zone. The highest level of nitrogen is also double maximum nitrogen uptake, since animal waste tends to be over-applied to field at an animal-crop operation.

$\overline{EC}$ [dS/m] is a critical value of soil salinity at which crop yield decreases to zero. It is calculated from the salt tolerance parameters of the crop. Maas and Hoffman [1977] specified the relationship between relative yield and soil salinity as Equation 4.2. When the soil salinity EC is greater than the threshold value A [dS/m], crop yield starts to decrease. The slope B expresses yield depression per dS/m. For many crops, both values of A and B are available in Maas and Hoffman [1977]. Setting relative yield to zero, $\overline{EC}$ can be calculated as shown in Equation 4.3.

$$ry = 1 - 0.01B\,(EC - A) \tag{4.2}$$

$$\overline{EC} = A + \frac{100}{B} \tag{4.3}$$

4.5 Crop Response Function Estimation

In this section, I demonstrate how I use the data presented in the previous section to fit the crop response functions. These functions provide crop uptakes (of water and nitrogen, which are required for the mass balance relationships and transition equations in the whole farm model), crop relative yield, and field emissions of nitrogen to the whole optimization problem and thus are very useful for policy simulations. Field emissions of drainage water and salt are calculated according to mass balance relationships in the root zone, which is demonstrated in Chapter 5.

The crop response functions are expressed as

$$wup = \Psi^{c}_{wup}(w, n, s) \tag{4.4}$$

$$nup = \Psi^{c}_{nup}(w, n, s) \tag{4.5}$$

$$ry = \Psi^{c}_{ry}(w, n, s) \tag{4.6}$$

$$nl = \Psi^{c}_{nl}(w, n, s) \tag{4.7}$$

w, n, and s are respectively crop available water [ha-cm/ha], available inorganic nitrogen [kgN/ha], and exposed salinity [dS/m]. See Chapter 5 for how to compute the three input factors during season k in time period t at field location j (i.e., $w_{t,k,j}$, $n_{t,k,j}$, and $s_{t,k,j}$), which need both soil dynamics and mass-balance relationships. wup, nup, ry, and nl are respectively water uptake [ha-cm/ha], nitrogen uptake [kgN/ha], crop relative yield, and nitrogen leaching [kgN/ha]. Ψ^c are functions for crop c that are estimated separately from the simulated dataset.

The forms of the uptake and relative yield functions are developed from the traditional Mitscherlich-Baule 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, I only present the latter in the following section. I also take corn as an example to illustrate the logic behind function modifications and the approximation procedure. Estimation results for the uptake functions of corn and the response functions of other crops are reported in Appendix C.

4.5.1 Relative Yield Function

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] compare 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 4.8, 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 4.9. 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.

$$Y = a \prod_i \left(1 - \exp\left(-b_i^1 \left(X_i + b_i^0\right)\right)\right) \tag{4.8}$$

$$ry \equiv \frac{y}{y^*} = \left(1 - \exp\left(-b_w^1 \left(w - b_w^0\right)\right)\right) \left(1 - \exp\left(-b_n^1 \left(n - b_n^0\right)\right)\right) \left(1 - \exp\left(-b_s^1 \left(s - b_s^0\right)\right)\right) \tag{4.9}$$

Unfortunately, Equation 4.9 fails to fit the data well. I find that, given a salinity level and a nitrogen level, the simulated yields have bell-shaped distributions over the full range of water levels (Figure 4.5). Therefore, I 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 4.5, the bell shapes vary for different salinity levels. Equation 4.9 is thus modified further so that salinity enters the function through its influence on the water and nitrogen parameters rather than directly as a separate multiplicative term.

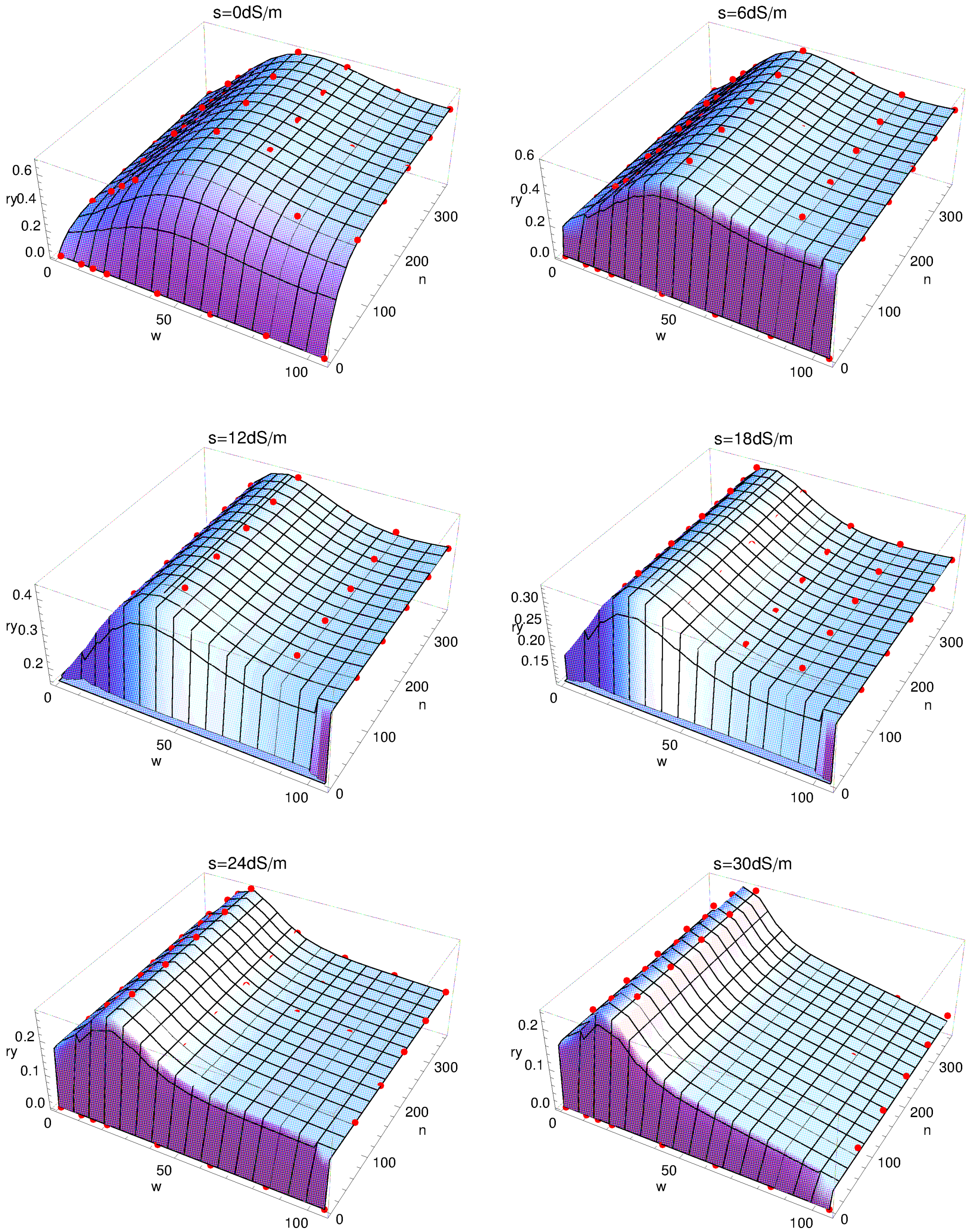

Figure 4.5: Relative yield vs. available water and available nitrogen when soil salinity is $0, 6, 12, 18, 24$, and 30 dS/m. Points: simulated data. Surfaces: fitted functions.

The relative yield function is thus defined as

$$ry = \left(1 - \exp\left(-b_w^1\left(\varphi w - b_w^0\right)\right)\right)\left(1 - \exp\left(-b_n^1\left(n - b_n^0\right)\right)\right) \tag{4.10}$$

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

Subproblem 1 Estimate Equation 4.11 once for each value of $s = 0, 6, 12, 18, 24,$ 30 dS/m, using the appropriate subset of simulated data points.

$$ry = g^{ry}(w, n, \mathbf{\Upsilon}) \tag{4.11}$$

See Equation 4.10 for the explicit form of g^{ry}. These estimations produce the surfaces shown in Figure 4.5. The figures show excellent agreement between simulated data (points) and fitted data (surfaces) at each salinity level.

Subproblem 2 Estimate each parameter $\Upsilon_i \in \mathbf{\Upsilon}$ as a polynomial function of salinity, as shown in Equation 4.12. Table 4.4 reports the estimated functions and Figure 4.6 depicts the regression curves. Again, agreement between the data (point estimates) and functions is generally very good.

$$\Upsilon_i = f_i^{ry}(s) \tag{4.12}$$

Substitute fitted Equation 4.12 into Equation 4.10 to get relative yield function Ψ_{ry}. This approach is verified by the good agreement between simulated data and the

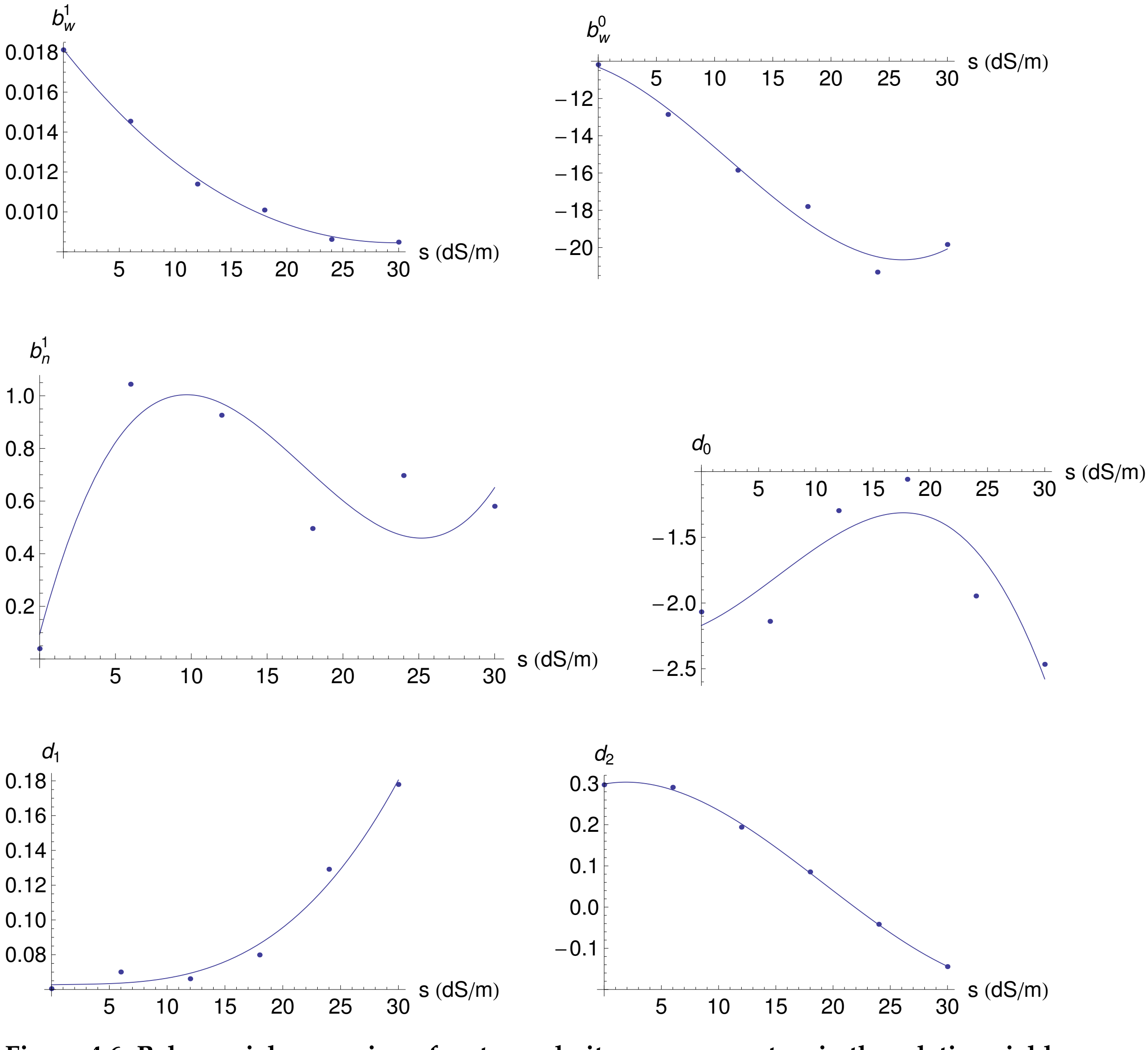

Figure 4.6: Polynomial regression of water and nitrogen parameters in the relative yield function

Table 4.4: 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	-7.08E-4	1.48E-5	-6.51E-8	0.9973
b_w^0	-10.3161	-0.2408	-0.0270	8.05E-4	0.9815
b_n^1	0.09396	0.2153	-0.0154	2.94E-4	0.7995
d_0	-2.1708	0.0398	0.0038	-1.84E-4	0.7736
d_1	0.0627	1.34E-4	-2.62E-5	-5.09E-6	0.9846
d_2	0.2987	0.0050	-0.0014	2.38E-5	0.9993

fitted relative yield from Ψ_{ry} (Figure 4.7).

$$ry = g^{ry}(w, n,\ f^{ry}(s)) = \Psi_{ry}^{corn}(w, n, s) \tag{4.13}$$

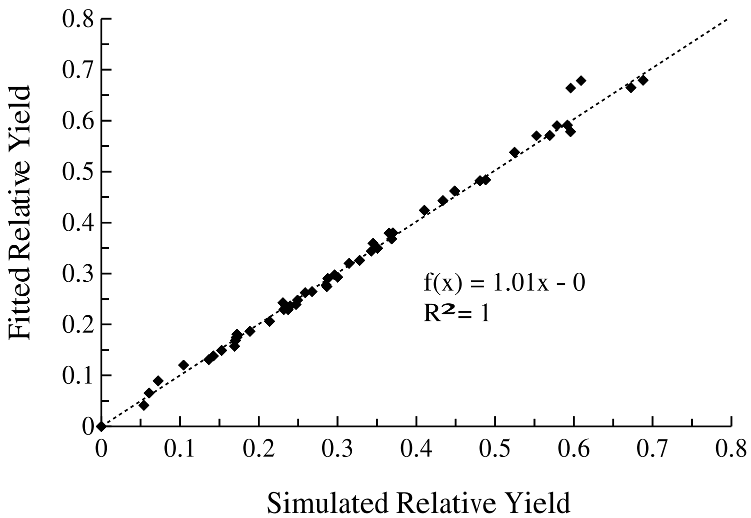

Figure 4.7: Relative yield function: simulated data vs. fitted data

4.5.2 Field Emission Function

I 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 4.14 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.

$$nl = \frac{\vartheta_n \cdot n}{1 + \exp\left(-\vartheta_w^1 \left(w - \vartheta_w^0\right)\right)} \tag{4.14}$$

$$\vartheta_i = f_i^{nl}\left(s\right) \tag{4.15}$$

I adopt the same procedure to estimate the field emission function. Equation 4.14 is estimated for $s = 0, 6, 12, 18, 24, 30$ dS/m (Figure 4.8). Equation 4.15 is then estimated $\forall \vartheta_i \in \boldsymbol{\vartheta}$ and substituted into Equation 4.14 to generate the field emission function $\Psi_{nl}(w, n, s)$ (Figure 4.9, Table 4.5). Figure 4.10 shows that the estimated field emission function fits the simulated data well.

In summary, the results show that my methodologies constitute a reliable approach to estimating the crop response functions with crop available water, available nitrogen, and exposed salinity as three input factors.

Table 4.5: Polynomial regression of water and nitrogen parameters in the field emission function

ϑ_i		$f_i^{nl}\left(s\right)$		
	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

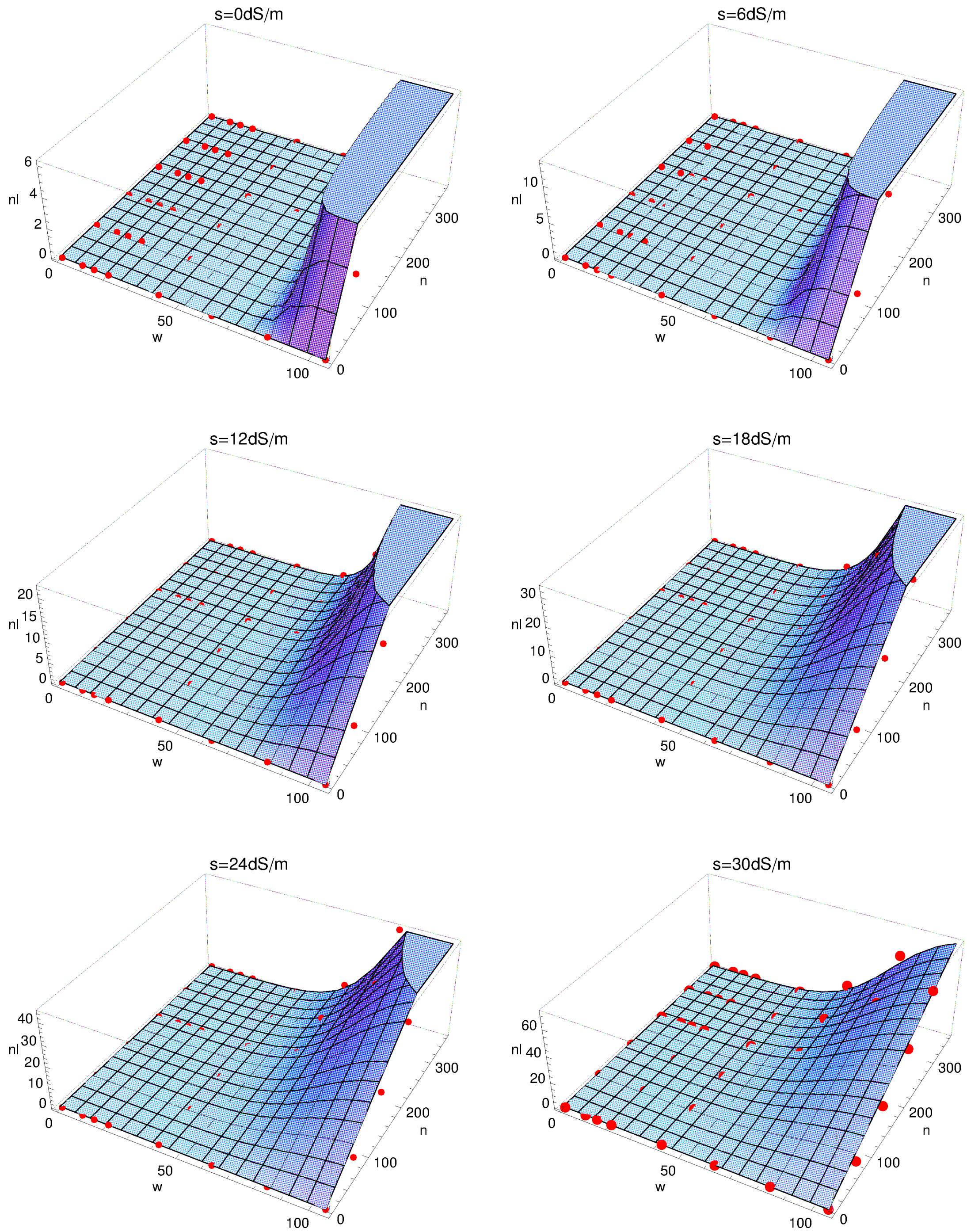

Figure 4.8: Field emission vs. available water and available nitrogen when soil salinity is $0, 6, 12, 18, 24$,and 30 dS/m. Points: simulated data. Surfaces: fitted functions.

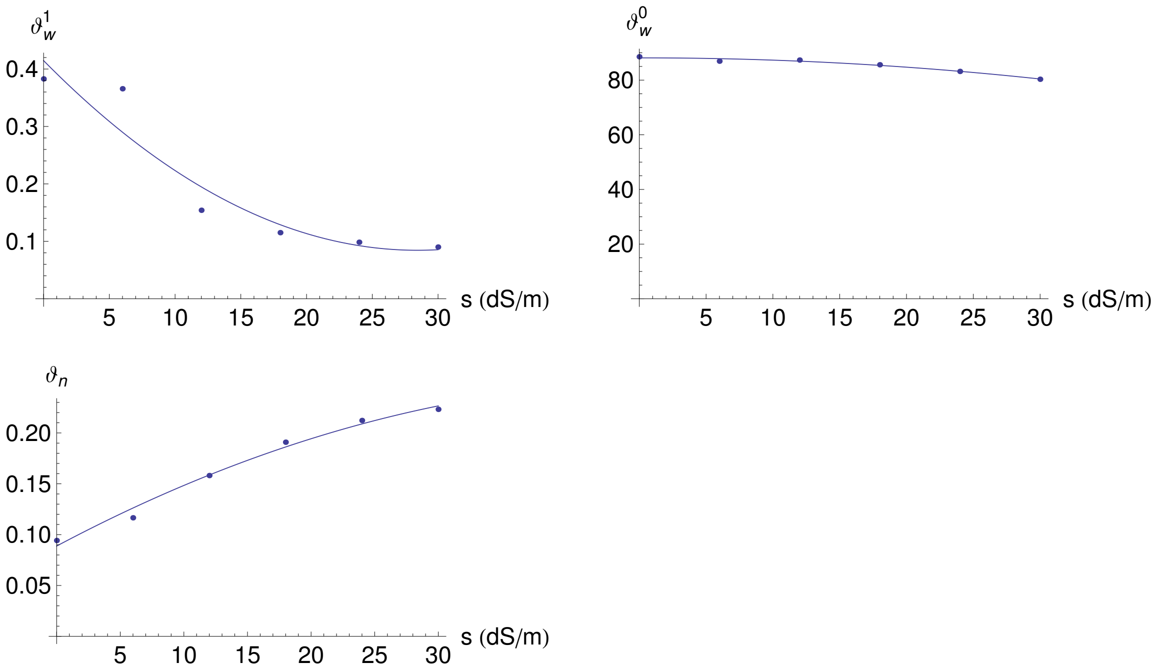

Figure 4.9: Polynomial regression of water and nitrogen parameters in the field emission function

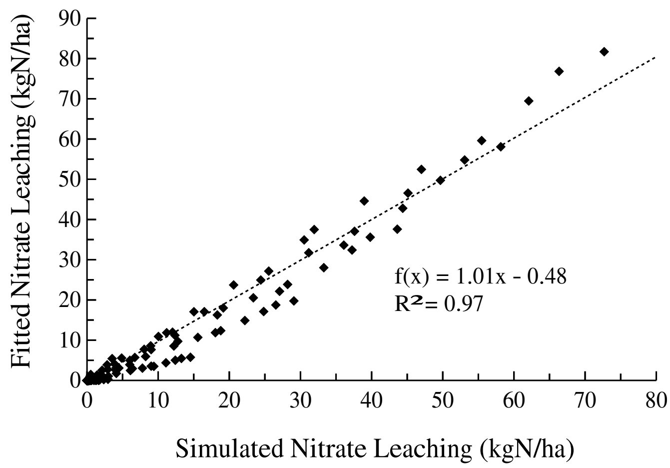

Figure 4.10: Field emission function: fitted data vs. simulated data

4.5.3 Alternative Function Specification

The crop response functions estimated above depend on absolute values of water [cm], nitrogen [kgN/ha], and salinity [dS/m]. As Letey and Dinar [1986] point out, for purposes of transferring such relationships among different geographical areas, it is helpful if these functions can be expressed in relative terms. I can achieve this by simple transformations.

Take the relative yield function as an example. Relative values are equal to absolute values divided by scaling factors, which are listed in Table 4.6. Transform the relative yield function in Equation 4.13 accordingly to get Equation 4.16, a new function Ω that only relies on relative terms. Since the scales $ET^*, n^*_{up},$ and $\overline{EC}$ are available for most regions, Ω can be transferred to the same type of crop grown in different regions.

$$
\begin{aligned}
ry \equiv G^{ry}(w,n,s) &\equiv G^{ry}\big(rw \cdot ET^*,\ rn \cdot n^*_{up},\ rs \cdot \overline{EC}\big) \\
&= \Omega\big(rw, rn, rs \mid ET^*, n^*_{up}, \overline{EC}\big) \qquad (4.16)
\end{aligned}
$$

Table 4.6: Scaling factors for calculating relative value

Absolute Value	Scale	Relative value
Available water w [cm]	Maximum Evapotranspiration ET^* [cm] (or pan evaporation)	$rw = \frac{w}{ET^*}$
Available nitrogen n [kgN/ha]	Maximum nitrogen uptake n^*_{up} [kgN/ha]	$rn = \frac{n}{n^*_{up}}$
Soil salinity s [dS/m]	Salinity critical value $\overline{EC}$ [dS/m]	$rs = \frac{s}{\overline{EC}}$
Yield y [ton/ha]	Maximum yield y^* [ton/ha]	$ry = \frac{y}{y^*}$

Chapter 5

Hydrologic Model

This chapter discusses subsurface flows and biogeochemical processes (i.e., nitrates/salts transportation and transformation) in both unsaturated and saturated zones that are relevant for the model farm. The unsaturated zone produces the three inputs for the crop response functions and generates field emissions, while the saturated zone takes the outputs from the unsaturated zone to determine downstream emissions into the groundwater aquifer.

Figure 5.1 demonstrates my conceptual modeling framework. In addition to precipitation, the farm operator can obtain water from three sources: surface water (imported via canals), deep groundwater (old groundwater in the deep aquifer that is not affected by farm effluent in the short term), and shallow groundwater (drainage water that can be recycled via capture wells at the downstream end of the farm). The quality and costs of surface water and deep groundwater are constant. The basic assumption is that, for a single farm model, changes of the ground water table and chemical concentrations in the deep aquifer are neglected. Another way to look at this assumption is a hypothetical aquitard (e.g., clays and shales) that divides the saturated zone into an upper local aquifer and a lower regional aquifer. The capture wells are relatively shallow and capture essentially only the farm's drainage water from the root zone. Drainage water that is not captured and recycled in the next time period enters the deep aquifer, carrying nitrates and salts as downstream emissions.

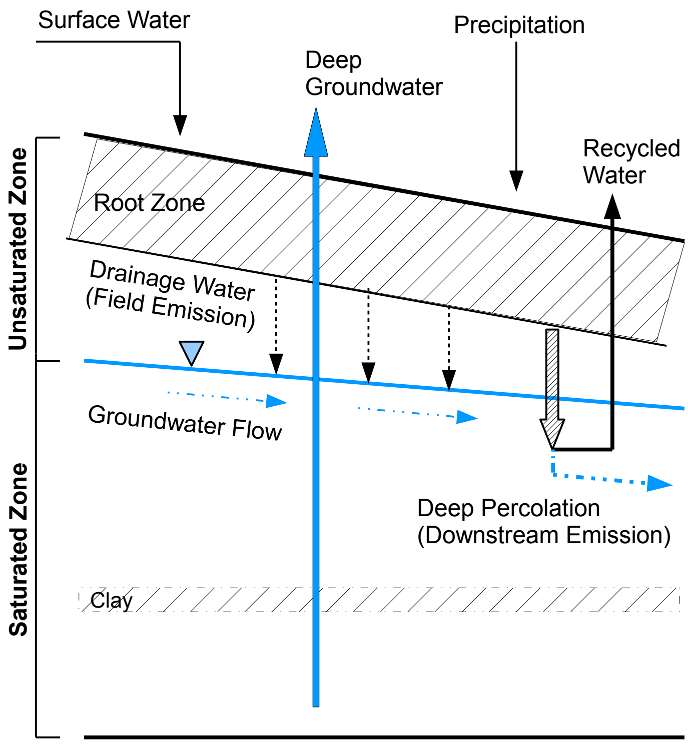

Figure 5.1: Conceptual hydrologic model

5.1 Unsaturated Zone

5.1.1 Soil Dynamics and Root Zone Supply

Soil dynamics are characterized by three state variables: soil inorganic nitrogen [kgN/ha], soil organic nitrogen [kgN/ha], and soil salinity [dS/m]. With additional field inputs, the root zone supplies water and chemicals to crops.

Water Equation 5.1 specifies crop available water as the weighted sum of five potenital sources of water that might be applied to field: surface water ($sw_{t,k}$), deep groundwater ($gw_{t,k}$), recycled water ($rw_{t,k}$), lagoon water ($l_{t,k}/\mu^M$), and precipitation ($prec_{t,k}$). Surface water, deep groundwater, and recycled water are mixed and delivered to the field via the irrigation system, leading to the spatial heterogeneity of water infiltration (β_j^I). Whether animal waste is uniformly applied depends on the type of manure handling system ($\beta^M = \beta_j^I$ for the flush-lagoon system and $\beta^M = 1$ for the scrape-tank system). Soil moisture is relatively low compared to the amount of applied water, and thereby is assumed way from water available to crops [Letey and Knapp, 1995].

$$w_{t,k,j} = \beta_j^I \left(sw_{t,k} + gw_{t,k} + rw_{t,k} \right) + \beta^M l_{t,k}/\mu^M + prec_{t,k} \tag{5.1}$$

Nitrogen Equation 5.2 specifies crop available inorganic nitrogen as a function of soil inorganic nitrogen ($in_{t,k,j}^{soil}$), soil organic nitrogen ($on_{t,k,j}^{soil}$) , solid ($sol_{t,k}$) and liquid ($l_{t,k}$) waste nitrogen applied onsite, total nitrogen from irrigation water ($n_{t,k,j}^w$), the amount of commercial fertilizer applied ($fl_{t,k}$) [kgN/ha], and the rate of atmospheric nitrogen deposition ($ad_{t,k}$) [kgN/ha]. δ_k is the seasonal mineralization rate of organic nitrogen, ϕ is the fraction of applied liquid waste nitrogen that volatilizes during application, and ω is the fraction of the organic nitrogen in lagoon water.

$$\begin{aligned} n_{t,k,j} = in_{t,k,j}^{soil} + \delta_k \left(on_{t,k,j}^{soil} + sol_{t,k} + \beta^M \omega \, l_{t,k} \right) \\ + \beta^M (1-\phi)\,(1-\omega)\, l_{t,k} + n_{t,k,j}^w + fl_{t,k} + ad_{t,k} \end{aligned} \tag{5.2}$$

The dynamics of soil organic and inorganic nitrogen closely follows Baerenklau et al. [2008], as shown in Equation 5.3 and 5.4. λ_k accounts for seasonal denitrification in the unsaturated zone. Initial values $on^{soil}_{1,1,j}$ and $in^{soil}_{1,1,j}$, $j = 1, \ldots, J$, are given.

$$\begin{cases} on^{soil}_{t,k+1,j} = (1 - \delta_k) \left(on^{soil}_{t,k,j} + sol_{t,k} + \beta^M \omega \, l_{t,k} \right), \\ \qquad t = 1, \ldots, T;\ k = 1, \ldots, K-1;\ j = 1, \ldots, J \\ on^{soil}_{t,1,j} = (1 - \delta_K) \left(on^{soil}_{t-1,K,j} + sol_{t-1,K} + \beta^M \omega \, l_{t-1,K} \right), \\ \qquad\qquad t = 2, \ldots, T+1;\ j = 1, \ldots, J \end{cases} \tag{5.3}$$

$$\begin{cases} in^{soil}_{t,k+1,j} = (1 - \lambda_k) \, n_{t,k,j} - nup^R_{t,k,j} - nl^R_{t,k,j}, \\ \quad t = 1, \ldots, T;\ k = 1, \ldots, K-1;\ j = 1, \ldots, J \\ in^{soil}_{t,1,j} = (1 - \lambda_K) \, n_{t-1,K,j} - nup^R_{t-1,K,j} - nl^R_{t-1,K,j}, \\ \qquad\qquad t = 2, \ldots, T+1;\ j = 1, \ldots, J \end{cases} \tag{5.4}$$

In Equation 5.2, the total amount of nitrogen from irrigation water ($n^w_{t,k,j}$) is determined by

$$\begin{cases} n^w_{t,k,j} = \beta^I_j \left(sw_{t,k} \, nc^{sw}_{t,k} + gw_{t,k} \, nc^{gw}_{t,k} + rw_{t,k} \, nc^{dw}_{t,k-1} \right) + prec_{t,k} \, nc^{prec}_{t,k}, \\ \qquad\qquad t = 1, \ldots, T;\ k = 2, \ldots, K,\ j = 1, \ldots, J \\ n^w_{t,1,j} = \beta^I_j \left(sw_{t,1} \, nc^{sw}_{t,1} + gw_{t,1} \, nc^{gw}_{t,1} + rw_{t,1} \, nc^{dw}_{t-1,K} \right) + prec_{t,1} \, nc^{prec}_{t,1}, \\ \qquad\qquad t = 1, \ldots, T;\ j = 1, \ldots, J \end{cases} \tag{5.5}$$

where $nc^a_{t,k}$ denote the nitrogen concentration [kgN/ha-cm] of water source a, $a \in \{sw, gw, rw, prec\}$. Set $rw_{1,1}$ to be zero.

Salt A few models have been developed for predicting salt concentrations in the root zone under intraseasonal irrigation [Bresler, 1967, Knapp, 1984]. Given my inter-seasonal modeling framework, I specify a relatively simple model for the dynamics of soil salinity

($s^{soil}_{t,k,j}$), as shown in Equation 5.6. During irrigation events, soil moisture content can be replenished up to the saturated percentage; after a certain time, the water stops moving and soil-moisture content remain constant at field capacity (ν) [Bresler, 1967]. Salt concentration of soil solution equals the total amount of salt (in the soil and from irrigation) divided by the total amount of water that is not taken up by the crop. Initial values $s^{soil}_{1,1,j}$, $j = 1, \ldots, J$, are given. Note that $s^{soil}_{t,k+1,j}$ is equivalent to $s_{t,k,j}$ in Equation 4.4-4.7.

$$\begin{cases} s^{soil}_{t,k+1,j} = \dfrac{\nu\, s^{soil}_{t,k,j} + s^{w}_{t,k,j}}{\nu + w_{t,k,j} - wup^{R}_{t,k,j}}, & t = 1, \ldots, T;\ k = 1, \ldots, K-1;\ j = 1, \ldots, J \\ s^{soil}_{t,1,j} = \dfrac{\nu\, s^{soil}_{t-1,K,j} + s^{w}_{t-1,K,j}}{\nu + w_{t-1,K,j} - wup^{R}_{t-1,K,j}}, & t = 2, \ldots, T+1;\ j = 1, \ldots, J \end{cases} \tag{5.6}$$

The total salt mass from irrigation water ($s^{w}_{t,k,j}$) in Equation 5.6 is determined by

$$\begin{cases} s^{w}_{t,k,j} = \beta^{I}_{j} \left(sw_{t,k}\, ec^{sw}_{t,k} + gw_{t,k}\, ec^{gw}_{t,k} + rw_{t,k}\, ec^{dw}_{t,k-1} \right) + \beta^{M} l_{t,k}\, ec^{l}_{t,k}/\mu^{M} + prec_{t,k}\, ec^{prec}_{t,k}, \\ \qquad t = 1, \ldots, T;\ k = 2, \ldots, K,\ j = 1, \ldots, J \\ s^{w}_{t,1,j} = \beta^{I}_{j} \left(sw_{t,1}\, ec^{sw}_{t,1} + gw_{t,1}\, ec^{gw}_{t,1} + rw_{t,1}\, ec^{dw}_{t-1,K} \right) + \beta^{M} l_{t,1}\, ec^{l}_{t,1}/\mu^{M} + prec_{t,1}\, ec^{prec}_{t,1}, \\ \qquad t = 1, \ldots, T;\ j = 1, \ldots, J \end{cases} \tag{5.7}$$

where $ec^{a}_{t,k}$ denote electrical conductivity [dS/m] of water source a. Initial value $ec^{dw}_{0,K}$ is given. Since $rw_{1,1}$ is set to be zero, $ec^{dw}_{0,K}$ can be given an arbitrary value (or assign $ec^{dw}_{0,K}$ a large value and then the optimal result for $rw_{1,1}$ will be zero).

Crop available water, available inorganic nitrogen, and exposed salinity, as specified in Equation 5.1, Equation 5.2, and Equation 5.6, act as the three input factors to the crop response functions (Equation 4.4-4.7).

5.1.2 Drainage

Water The volume of water drained from a unit area ($dw_{t,k}$) [cm/ha] is equal to total applied water less crop water uptake. Because of non-uniform irrigation systems, the value is a weighted summation over the J subareas of the field, which is same for all the other outputs.

$$dw_{t,k} = \sum_{j=1}^{J} pr\beta_j^I \left(w_{t,k,j} - wup_{t,k,j}^R\right)$$

Nitrogen Field emission of nitrogen ($fe_{t,k}^n$) per unit area [kgN/ha] is specified as

$$fe_{t,k}^n = \sum_{j=1}^{J} pr\beta_j^I \, nl_{t,k,j}^R$$

Nitrate concentration of drainage water ($nc_{t,k}^{dw}$) [kgN/ha-cm] is then determined by

$$nc_{t,k}^{dw} = \frac{fe_{t,k}^n}{dw_{t,k}}$$

Salt Field emission of salt per unit area ($fe_{t,k}^s$) [cm-dS/m] is specified as

$$fe_{t,k}^s = \sum_{j=1}^{J} pr\beta_j^I \, dw_{t,k,j} \, s_{t,k,j}$$

The salinity level of drainage water ($ec_{t,k}^{dw}$) [dS/m] is then determined by

$$ec_{t,k}^{dw} = \frac{fe_{t,k}^s}{dw_{t,k}}$$

The nitrate concentration and the salinity level of drainage water (or recycled water) enter Equation 5.5 and Equation 5.7 respectively. Drainage water, field emission of nitrogen, and field emission of salt enter the submodel for the saturated zone.

5.2 Saturated Zone

5.2.1 Nitrogen

After the nitrogen leaves the root zone, its transport to the water table is complicated by the various nitrogen species and transformations that can occur in the unsaturated zone and saturated zone. The forms and concentrations of nitrogen that reach the groundwater aquifer are quite site-specific, depending on interactions of physical and microbial processes such as filtering of particulate organic nitrogen, microbial assimilation, ammonium sorption, denitrification, and ammonia volatilization [DeSimone and Howes, 1998]. There have been great efforts in the literature trying to quantify the transport and transformation of nitrogen in groundwater aquifers. From a mass balance analysis, DeSimone and Howes [1998] conclude that ammonification and nitrification in the unsaturated zone and ammonium sorption in the saturated zone are predominant while loss of fixed nitrogen through denitrification is minor. Despite these findings in the hydrogeologic and hydrochemical literatures, all the studies of groundwater nitrogen pollution in the economics literature have only accounted for the reactions in root zone and unsaturated zone, mainly nitrification-denitrification and mineralization of organic nitrogen. Therefore, the nitrate attenuation capacity of the saturated zone, together with potential cost-saving management practices, is missing from previous economic analyses of nitrogen pollution of groundwater.

The processes of ammonification, nitrification, and denitrification in the unsaturated zone are characterized by the parameters $\phi, \delta,$ and λ in Chapter 5.1.1. For the saturated zone, ammonium sorption and denitrification need to be characterized and incorporated into my model. According to DeSimone and Howes [1998], in the case of a plume of septage-contaminated groundwater in Massachusetts, ammonium sorption in the saturated zone can remove up to 16 percent of the recharged nitrogen mass from the groundwater. When applied to the case of land application of dairy manure, adjustments need to be made depending on field conditions. What is more difficult to determine is whether and how to account for denitrification in the saturated zone. A closer look at DeSimone and Howes [1998]'s discussion can be helpful: although denitrification in the anoxic zone reduced only about 2 percent of the recharged nitrogen mass, they explain that denitrification is limited by organic carbon concentrations in the aquifer. Otherwise the nitrogen mass removed through denitrification will continue to increase, which is indicated by the expanding anoxic zone in the study site. This coincides with the modeling results of Lee et al. [2006] for a field study at a cattle feedlot site in Western Australia, where sawdust is used as an additional carbon source to enhance microbial denitrification. Recently Singleton et al. [2007] do find, through their field studies at two dairy operations in the Central Valley of California, that saturated zone denitrification can mitigate the impact of nitrate loading at dairy operations, especially when lagoon seepage provides a source of carbon (local capture wells may intensify this supply by inducing significant lateral flow) and thus increases the likelihood of denitrification. Although there are many uncertainties, dairy operations seem to establish conditions conducive to saturated zone denitrification.

5.2.2 Salts

When one thinks about salt in dairy manure, it's necessary to distinguish between nutrient salinity and non-nutrient salinity. The major cations in typical irrigation waters are Ca^{+}, Mg^{2+}, and Na^{+}, of which only a small portion are absorbed by plants, while dairy lagoon waters also have high concentrations of NH_4^+ and K^+that plants can take up in larger quantities as nutrients [Chang et al., 2005]. If the nutrient part of salinity (NH_4^+ and K^+) is removed by the crop, it has no impact on crop growth as in typical salinity case. Analyses of sampled dairy lagoon water in Central Valley of California reveal that on average the nutrient salinity of the lagoon water is approximately 58 percent of what the measured EC indicates [Chang et al., 2005].

Salts leaching from the root zone will finally reach groundwater and lead to groundwater salinization. Through the recycling of wastewater by local pumping, nitrogen can be circulated and reused. However, salt is a "conservative" pollutant, which is not active like nitrogen in soil or groundwater. In the short term, this is not a severe problem for irrigated agriculture since in some areas saline drainage water is reused and many studies have demonstrated the feasibility of this management practice [Jury et al., 1978, Rhoades, 1989, Schwabe et al., 2006]. The long term impacts are still not well understood. However, due to the high concentration of salts in manure, it's possible that even a small amount of recycling water at a dairy operation eventually would turn the location into a hot-spot of salinity.

5.2.3 Mass Balance Relations

There are various ways to model the groundwater flow and solute transport/ transformation in the saturated zone, such as hydraulic transient models [Harbaugh, 2005] and concentration response matrices [Peña-Haro et al., 2009]. In this thesis, I take a mass balance approach.

Drainage water that is not recycled enters the aquifer as deep percolation ($dp_{t,k}$) [cm/ha].

$$\begin{cases} dp_{t,k} = dw_{t,k} - rw_{t,k+1}, & t = 1, \ldots, T;\ k = 1, \ldots, K-1 \\ dp_{t,K} = dw_{t,K} - rw_{t+1,1}, & t = 1, \ldots, T \end{cases}$$

Downstream emission of nitrogen per unit area ($de^n_{t,k}$) [kgN/ha] is specified as

$$\begin{cases} de^n_{t,k} = \frac{dp_{t,k}}{dw_{t,k}} fe^n_{t,k} \left(1 - \left(1 + \frac{rw_{t,k+1}}{dw_{t,k}}\right) \alpha^{denitr} - \alpha^{miscel}\right), & t = 1, \ldots, T;\ k = 1, \ldots, K-1 \\ de^n_{t,K} = \frac{dp_{t,K}}{dw_{t,K}} fe^n_{t,K} \left(1 - \left(1 + \frac{rw_{t+1,1}}{dw_{t,K}}\right) \alpha^{denitr} - \alpha^{miscel}\right), & t = 1, \ldots, T \end{cases}$$

where α^{denitr} and α^{miscel} denote the nitrate loss rates due to denitrification and the other transformation processes. The additional coefficient for α^{denitr} stands for the intensified denitrification when more shallow groundwater is pumped.

Downstream emission of salt per unit area ($de^s_{t,k}$) [cm-dS/m] is specified as

$$de^s_{t,k} = \frac{dp_{t,k}}{dw_{t,k}} fe^s_{t,k}$$

In this thesis, field emission and downstream emission of nitrogen are the two main policy targets, for which I simulate and analyze different policy scenarios in Chapter 8.2 and Chapter 8.3. I leave the control of salt emission for future research.

Chapter 6

Whole Farm Economic Model

The objective function is specified as

$$\max_{\{\theta_{t,g},l_{t,k},sol_{t,k},sw_{t,k},gw_{t,k},rw_{t,k},fl_{t,k},M,I,R\}} \left[\sum_{t=1}^{T} \left[\eta^t \left(\pi_t^{herd} + \pi_t^{waste} + \pi_t^{crop} - \pi_t^{policy} \right) \right] + \eta^T \mathbf{p}^{\mathbf{herd}} \left(\boldsymbol{\zeta}_1 h_{T,G} \right)^\top \right] \quad (6.1)$$

subject to transition equations for the state variables, mass balance requirements, non-negativity constraints, herd permit limits, and policy-related constraints. Net farm income equals the net revenues from herd production, waste disposal, and crop production less the environmental policy costs (π_t^{policy}). η is the discount rate. The last term represents the salvage value of the herd. All the other variables have been defined previously.

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.

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

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, I 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). I 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 the possibility of switching in the future. At the beginning of each year in the planning period, the operator decides how many cows to buy or sell ($\theta_{t,g}$), and the amount of liquid animal waste ($l_{t,k}$), solid animal waste ($s_{t,k}$), surface water ($sw_{t,k}$), deep groundwater ($gw_{t,k}$), shallow groundwater ($rw_{t,k}$) and commercial fertilizers

($fl_{t,k}$) to be applied to the field for each season (k) within that year.

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. I already have a high dimensional model with several nonlinearities, so I 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. I 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 I denote the activities associated with the optimal solution by "the optimal activities").

Part III

Policy Analysis

Chapter 7

Baseline Scenario

7.1 Study Site and Data

7.1.1 Climate and Soil

I collect data on the region of 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, 2007b]: bulk density $1.5\,\mathrm{g/cc}$ (range $1.35 - 1.70\,\mathrm{g/cc}$), saturated hydraulic conductivity $28\,\mu\mathrm{m/sec}$ (range $14 - 42\,\mu\mathrm{m/sec}$) [USDA, 2009]. According to the soil survey in USDA [2009], the highest soil salinity in this region is around $20\,\mathrm{dS/m}$, which is well covered by my selected intervals of soil salinity for the crop response functions.

I assume a discount rate of 4 percent with all economic data inflation-adjusted to 2005 dollars.

7.1.2 Crop

Data on production fixed cost and maximum yield is from the University of California Cooperative Extension (Cost & Return Studies by UC Davis). Crop prices are from the National Agricultural Statistics Service.

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

Water Source The farmer can import high quality surface water ($EC = 0.15\,\text{dS/m}$ and $N = 1\,\text{mg/l}$) with a price of $\$2.58/\text{ha-cm}$ [Vargas et al., 2003], pump deep groundwater ($EC = 1.18\,\text{dS/m}$ and $N = 10\,\text{mg/l}$) at a cost of $\$8.84/\text{ha-cm}$ [Marques et al., 2003], and pump shallow groundwater through capture wells at a cost of $\$2.58/\text{ha-cm}$ [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 $\$1.62/\text{ha-cm}$ to $\$11.35/\text{ha-cm}$ [Hutmacher et al., 2003]. Sensitivity analysis on surface water price is performed in Section 9.2.

Irrigation System 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. My 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 my empirical study focuses on the area of Tulare and surrounding counties, I only include $1/4$-mile furrow and linear move system in the simulations and analyses reported here.

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

All the parameters of the model are given in Appendix D.

7.2 Baseline Simulation (No Policy Implementation)

I 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 time period)[1]. Therefore, the following analyses are based on the results of the first 24 years. Table 7.1

[1]Here I use the term "steady state" in a broader sense than just a terminology associated with infinite deterministic dynamic programming problems. My model is a finite dynamic problem where the state variables can achieve steady levels.

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 total net farm income over the years is 26.24 million dollars, which will be used as a reference value to calculate the relative losses of net farm income under alternative policy scenarios. 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 7.1 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, as shown in Figure 7.2 , 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

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

Table 7.1: 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]

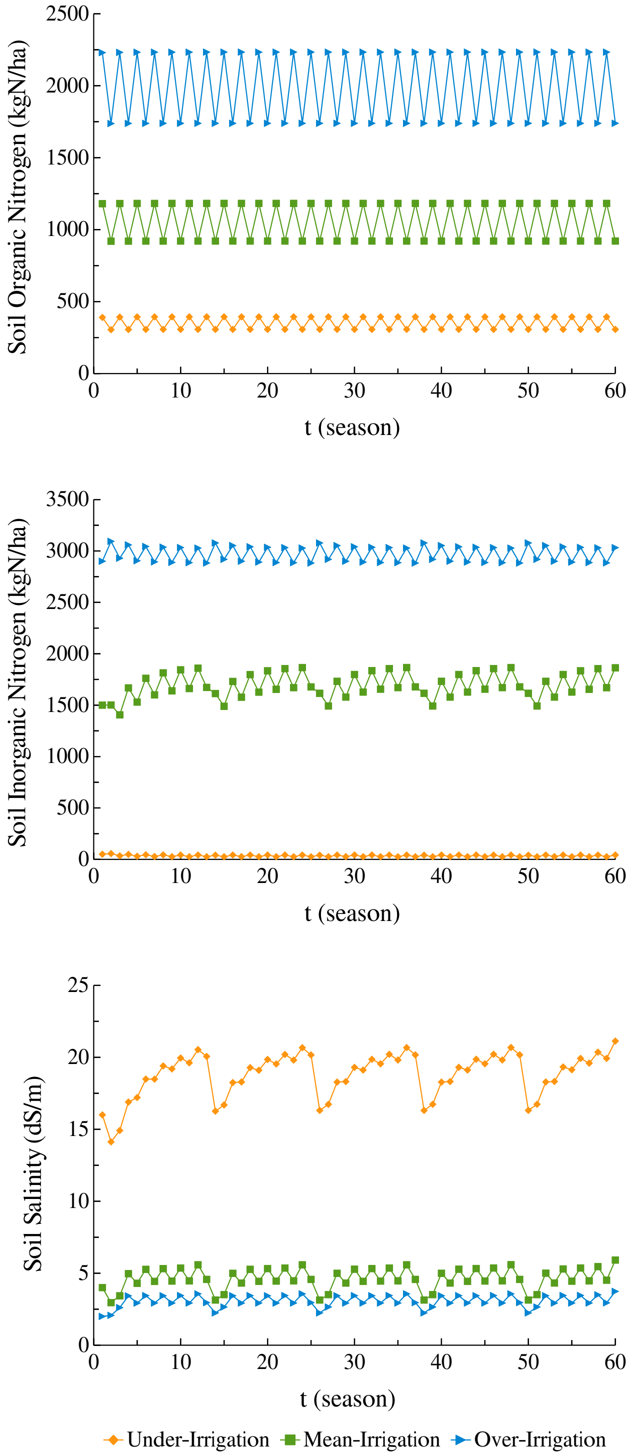

Figure 7.1: Baseline: paths of soil organic nitrogen, soil inorganic nitrogen, and soil salinity for each field type under the optimal activities (flush-lagoon, 1/4-mile furrow, corn-wheat rotation)

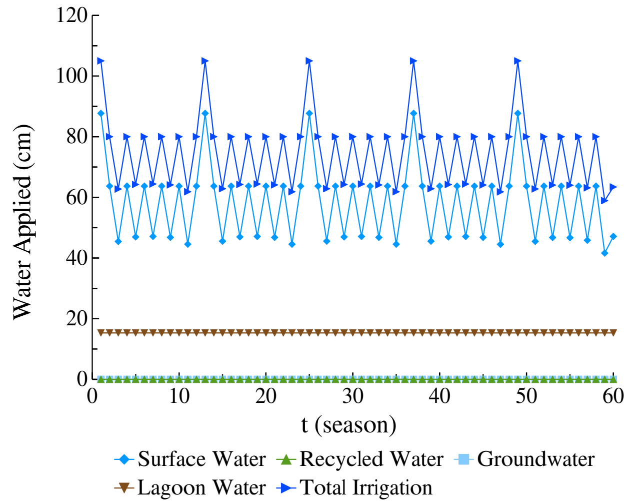

Figure 7.2: Baseline: paths of irrigations under the optimal activities (flush-lagoon, $1/4$-mile furrow, corn-wheat rotation), given three sources of irrigation water

organic nitrogen.

Figure 7.3 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 my simulated value and the comparison value for applied fertilizer in Table 7.1. I do not consider these issues for the dairy operator, but for surrounding land owners I use three levels of WTAM (20%, 60%, and 100%)[3] to account for these concerns and perform sensitivity analysis in Section 9.2. Furthermore I assume 25% of surrounding land is suitable for spreading animal waste [Baerenklau et al., 2008].

Figure 7.4 shows the water infiltration and field emission of nitrogen for each field

[3] All the results presented in this chapter and next chapter are for WTAM equal to 60%.

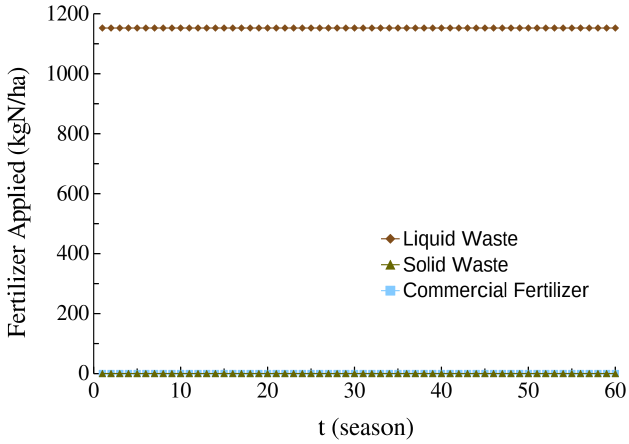

Figure 7.3: Baseline: path of fertilizer application under the optimal activities (flush-lagoon, $1/4$-mile furrow, corn-wheat rotation), given three sources of fertilizer

type. Flushing of salt also carries more nitrate 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 nitrate through the soil even if there is no flushing.

Table 7.2 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 4.2), gets 30.83% of total irrigation, produces 19.85% of total crop yield, but accounts for 77.19% of total field emission of nitrogen, which is why Knapp and Schwabe [2008] say that "nitrogen management is in a very real sense water management."

To further illustrate the effects of non-uniform irrigation, I report similar infor-

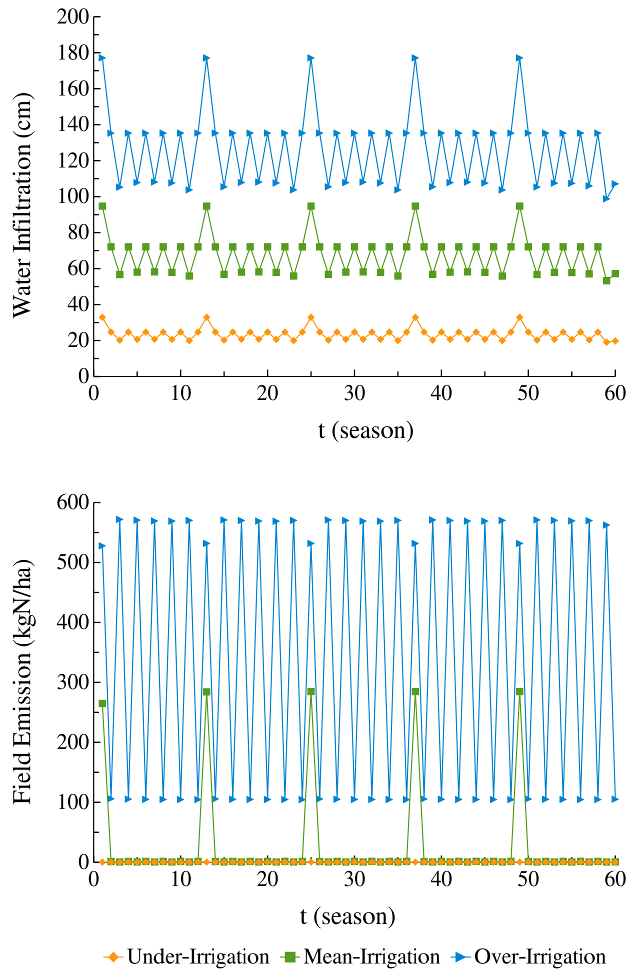

Figure 7.4: Baseline: paths of water infiltration and field emission for each field type under the optimal activities (flush-lagoon, $1/4$-mile furrow, corn-wheat rotation)

Table 7.2: 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 (cm)	80.81 [2.24%]	2415.80 [66.93%]	1112.69 [30.83%]	3609.29 [100.00%]
Relative Yield	1.51 [4.61%]	24.70 [75.54%]	6.49 [19.85%]	32.69 [100.00%]
Field Emission (kgN/ha)	0.97 [0.03%]	865.55 [22.78%]	2932.81 [77.19%]	3799.33 [100.00%]

mation in Table 7.3 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 7.5 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, the operator stops the periodic application of 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

Table 7.3: 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 (cm)	0.03 [0.00%]	2688.53 [78.93%]	717.66 [21.07%]	3406.22 (-*5.63%*) [100.00%]
Relative Yield	0.00 [0.00%]	29.16 [86.99%]	4.36 [13.01%]	33.52 (+*2.53%*) [100.00%]
Field Emission (kgN/ha)	0.00 [0.00%]	44.97 [2.18%]	2022.47 [97.82%]	2067.44 (-*45.58%*) [100.00%]

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 nitrogen field emission, but through different mechanisms. Figure 7.6 displays some optimal paths and decision rules under this alternative activity. The over-irrigated subfield has the smallest amount of both nitrogen and salt in soil because it has the highest level of leaching and because animal waste is uniformly applied under the scrape-tank system while mixed irrigation water is not (see Section 3.2.1). 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 7.1). 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 7.4 . 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

Table 7.4: 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%]

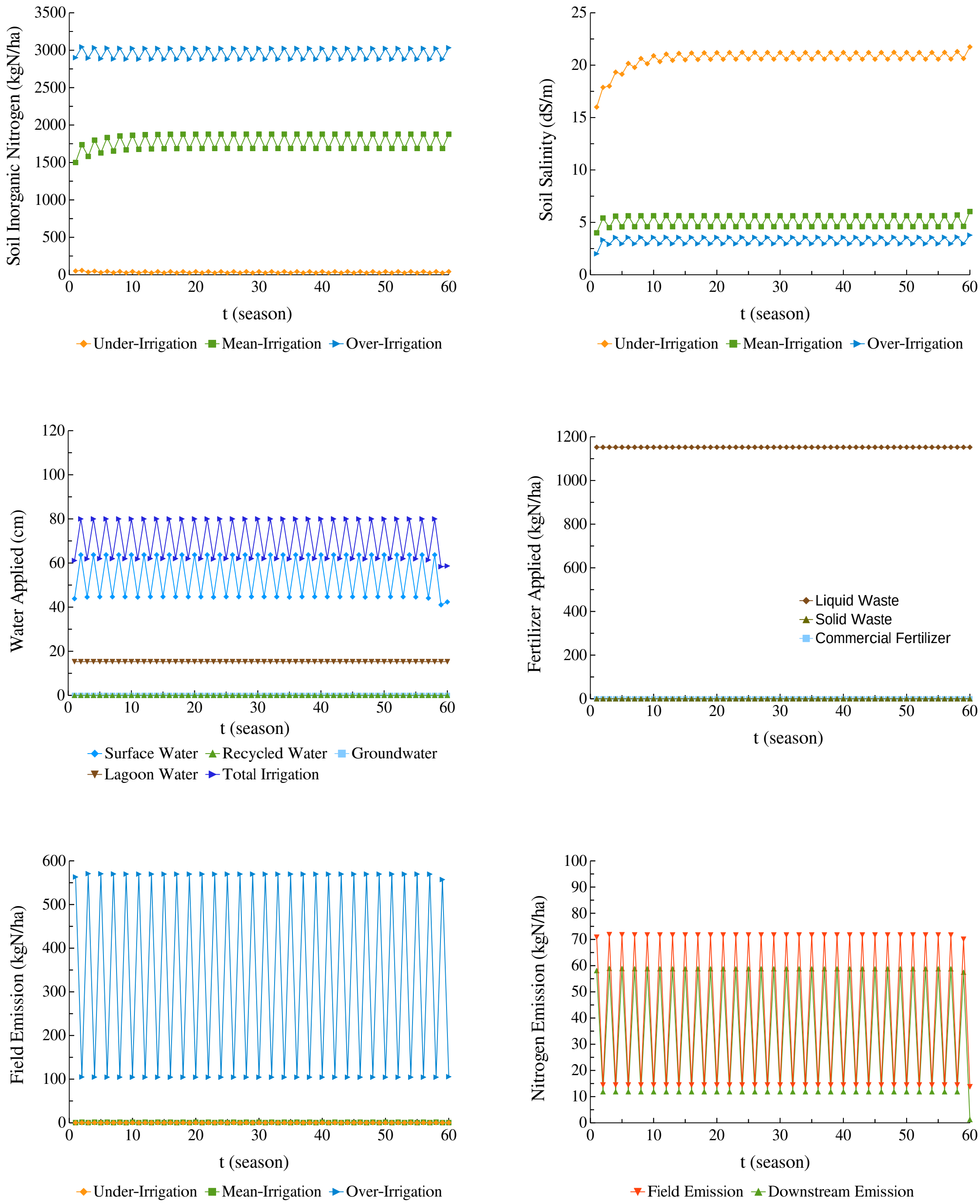

Figure 7.5: Baseline: paths of soil inorganic nitrogen, soil salinity, and field emission for each field type, paths of irrigation and fertilizer application, and seasonal nitrogen emissions under an alternative combination of activities (flush-lagoon, linear move, corn-wheat rotation)

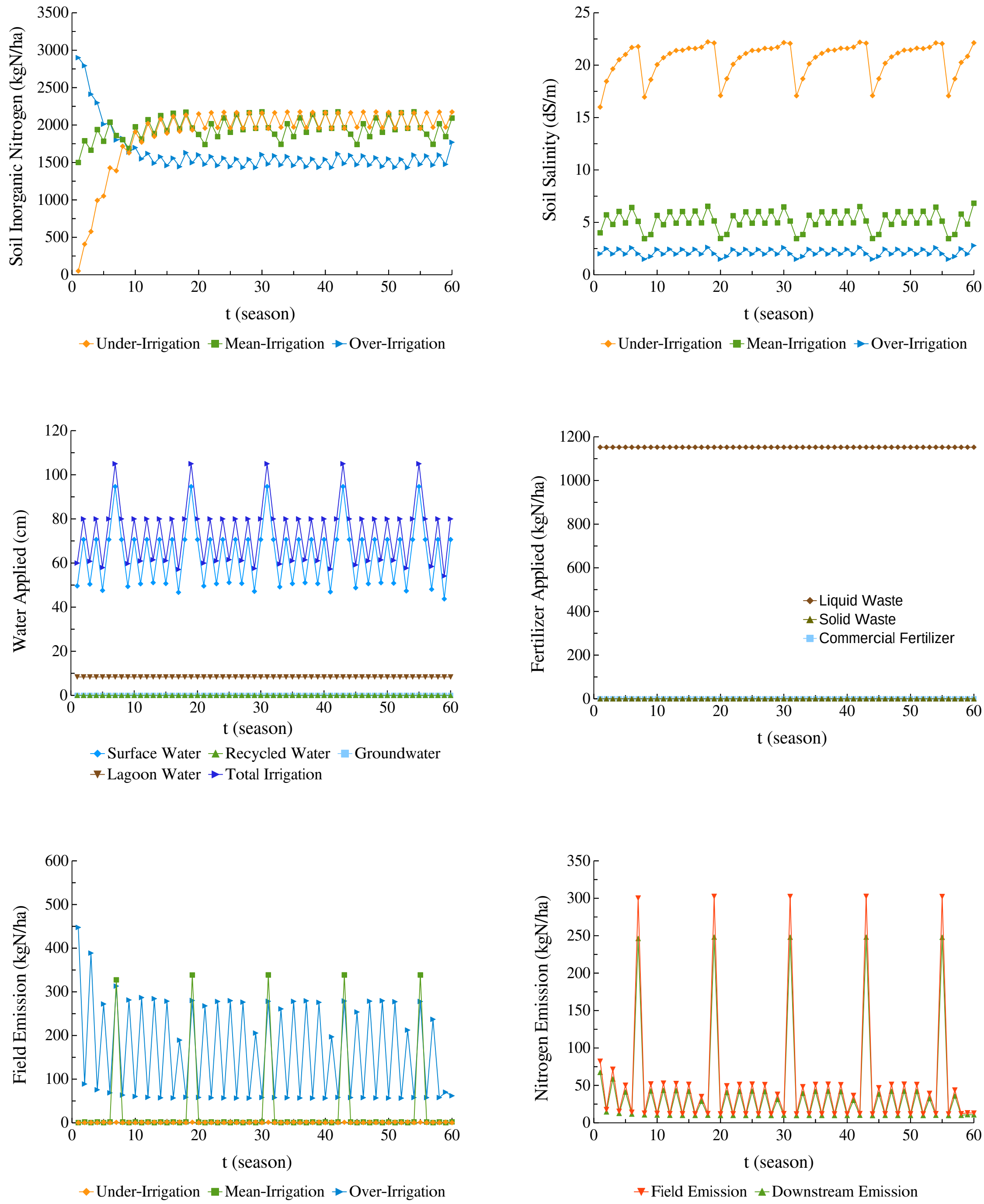

Figure 7.6: Baseline: paths of soil inorganic nitrogen, soil salinity, and field emission for each field type, paths of irrigation and fertilizer application, and seasonal nitrogen emissions under an alternative combination of activities (scrape-tank, $1/4$-mile furrow, corn-wheat rotation)

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 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.

Chapter 8

Alternative Policy Scenarios

Five alternative policy scenarios are simulated and analyzed in this thesis: NMPs, field emission limit (FEL), field emission charge (FEC), downstream emission limit (DEL), and downstream emission charge (DEC). NMPs is a quantity restriction on a contributing input. FEL and FEC are, respectively, a quantity restriction and emission charge on the intermediate pollution (i.e., the amount of pollutants emitted at an intermediate stage of their transport and transformation), while DEL and DEC are, respectively, a quantity restriction and emission charge on the final pollution (i.e., the net amount of pollutants that actually leave the property and enter the ecosystem). Therefore, the five policies span three levels of policy target and two types of control mechanism.

I am interested in the "life cycle" of a pollutant (i.e., different levels of pollution) because policies targeting the final pollution can be more cost-effective. The intuition is that these policies can create incentives for polluters to take into account more potential compliance strategies than policies that target inputs or intermediate pollutant emissions. This approach is similar to the widely used method of Life-Cycle Assessment in the sense that Life-Cycle Assessment tracks a commercial product while my analysis tracks a pollutant, both of which can help avoid a narrow outlook on environmental and economic concerns. Ideally, I need to link a physical-chemical-biological model with the evolution in space and time of the pollutant to an economic model. The whole farm model discussed in Part II is an attempt to achieve this goal.

The policy simulations are designed to evaluate the cost-effectiveness of alternative policies compared to NMPs: I use the field and downstream emissions from the optimal solutions under NMPs as reference points. To wit, I adjust the emission limits and charges under the FEL and FEC so that they achieve the same level of field emissions as achieved under the optimal NMPs; similarly, I adjust the emission limits and charges under the DEL and DEC to achieve the same downstream level of nitrogen loadings as achieved by the optimal solutions under NMPs[1]. The losses of net farm income are then compared. For each policy simulation, I assume the operation is initially at the steady state derived from the baseline scenario and then solve for the dynamically optimal practices under that policy.

8.1 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 my model, manure and process wastewater are equiva-

[1]Therefore, the field emissions from the optimal solutions under NMPs, FEL, and FEC are same, and the downstream emissions from the optimal solutions under NMPs, FEL, and FEC, DEL, and DEC are same.

lent to the animal waste, and nitrogen from previous crops is captured by soil dynamics. It is noted that the General Order requires soil tests from the land application area only once every 5 years for soluble phosphorus [CRWQCB, 2007]. Therefore, soil nitrogen is not accounted for by NMPs. As specified in Equation 8.1, the NMP requirement is included in the model as a constraint on seasonal application of inorganic nitrogen, which is the total of the nitrogen from applied animal waste (inorganic nitrogen plus the amount of organic nitrogen that is mineralized during that season), fertilizers, and irrigation water.

$$(1-\omega)\, l_{t,k} + \delta_k \left(sol_{t,k} + \omega\, l_{t,k} \right) + fl_{t,k} + n^{w}_{t,k} \leq 1.4\, n^{R*}_{up} \tag{8.1}$$

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 NMPs scenario. The main results are displayed in Figure 8.1 . Due to the NMP constraint, the operator hauls almost half of the liquid waste off site. This results in 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 7.6). 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.

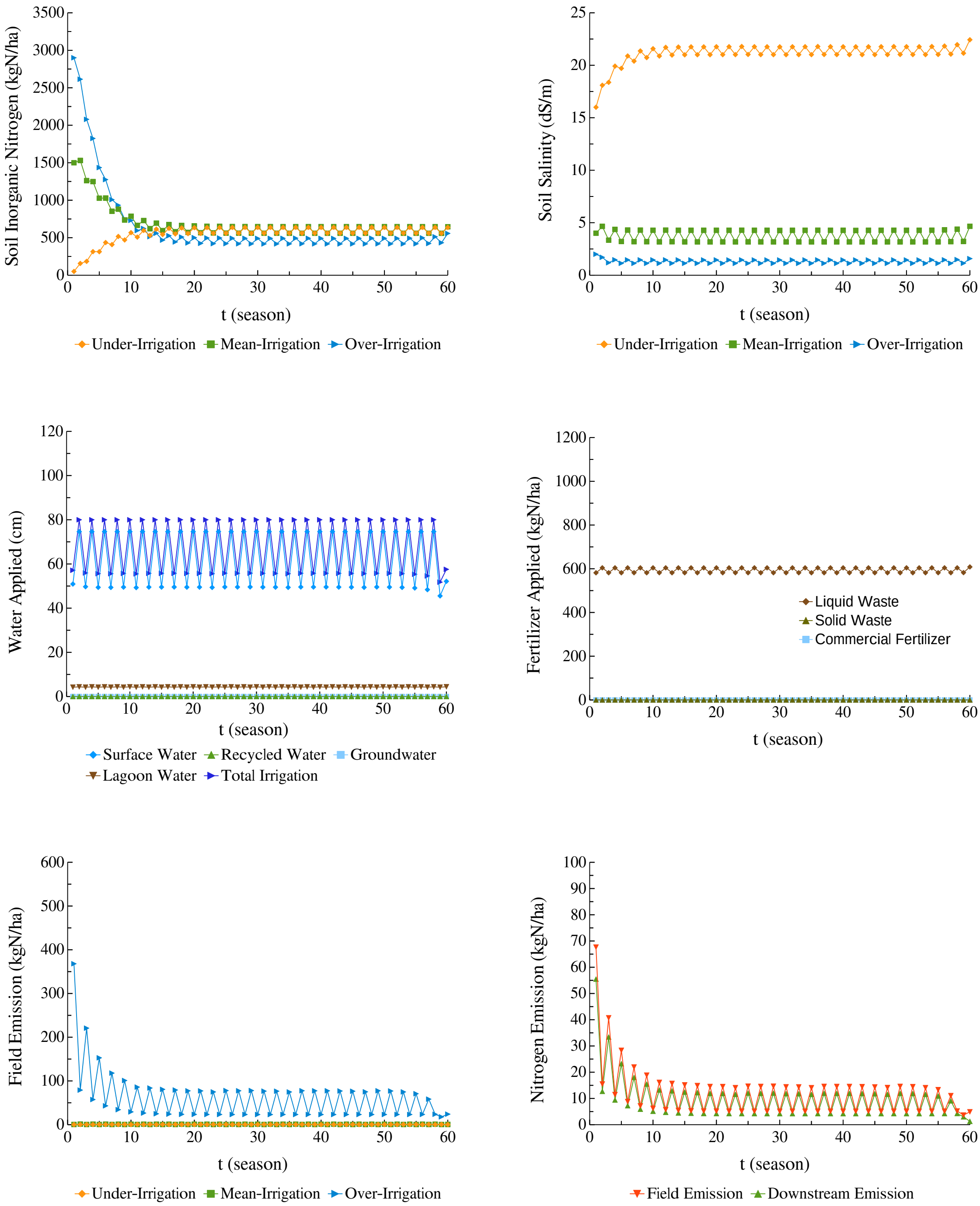

Figure 8.1: Nutrient Management Plans: paths of soil inorganic nitrogen, soil salinity, and field emission for each field type, paths of irrigation and fertilizer application, and seasonal nitrogen emissions under the optimal activities (scrape-tank, $1/4$-mile furrow, corn-wheat rotation)

8.2 Field Emission Control

8.2.1 Field Emission Limit

The total amount of field emissions over the 24 years horizon is $604.2\,\mathrm{kgN/ha}$, or approximately $25.2\,\mathrm{kgN/ha}$ per year, which is set as the annual cap on field emissions[2].

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 8.2 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. The first term on the right-hand side of Equation 5.4 indicates that 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 allows nitrate and salt to leach down to the aquifer, the operator takes advantage of the natural processes to reduce the field emissions of nitrogen.

The reduction in irrigation mainly happens in summer, since the winter crop is more salt-tolerant and under the baseline scenario field emissions from the summer cropping is five times more than that from the winter cropping (Figure 7.5). Less irrigation

[2] I also test a 6-year cap of $151.05\,\mathrm{kgN/ha}$ 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 flexibility provided by this longer time horizon. In practice, an emissions limit or charge would be imposed on estimated (or modeled) nitrogen emissions.

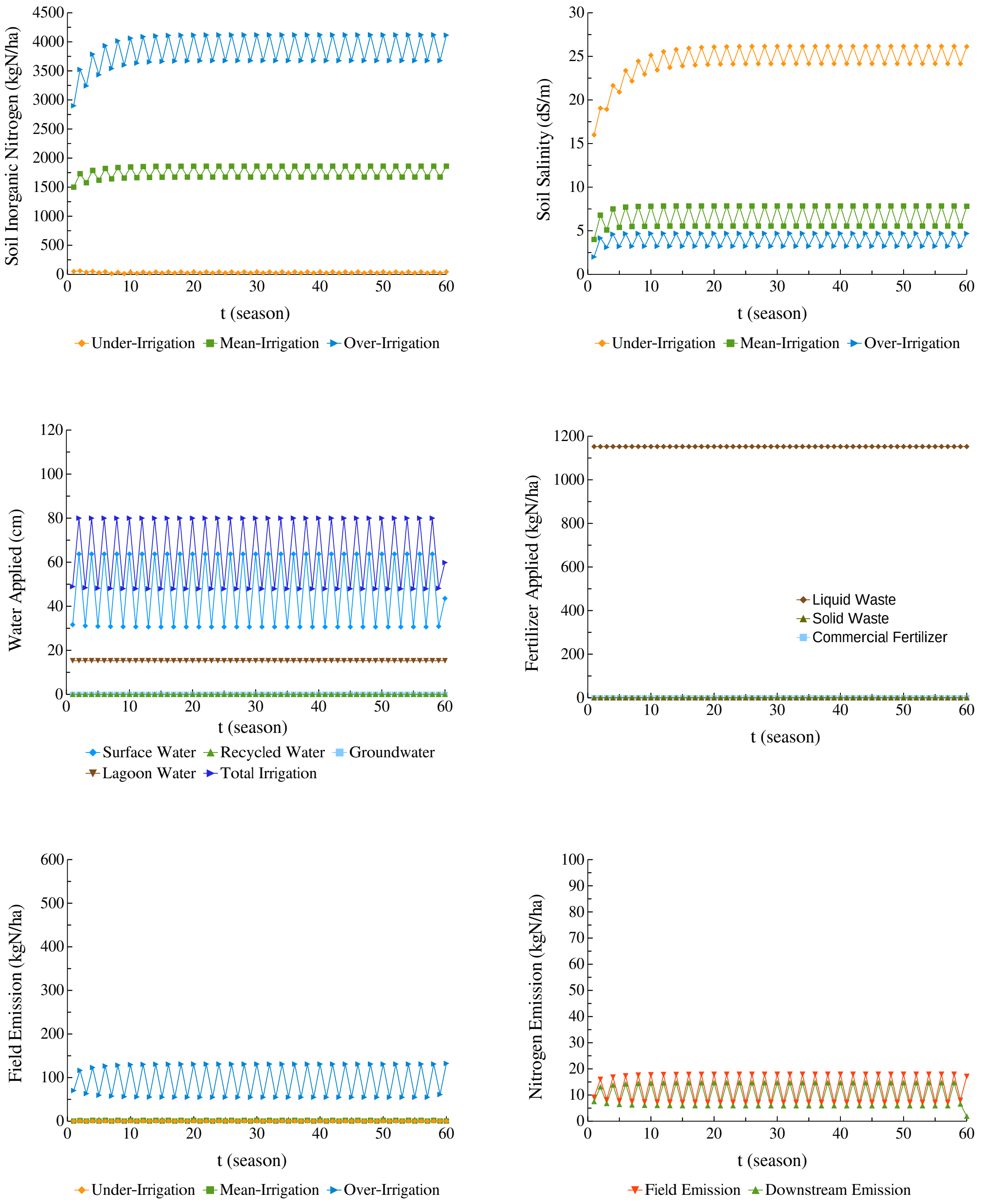

Figure 8.2: Field emission limit: paths of soil inorganic nitrogen, soil salinity, and field emission for each field type, paths of irrigation and fertilizer application, and seasonal nitrogen emissions under the optimal activities (flush-lagoon, linear move, corn-wheat rotation)

leads to higher levels of soil salinity in the subfields, which can reduce the crop yields. Compared to the optimal results of the baseline scenario, net farm income decreases by only 0.79%, with 7.20% of the crop revenue 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 emissions. The reasons are twofold: First, there is limited information on the relationships between the multiple inputs and the pollution, so it is difficult to determine the best approach of regulating the inputs (e.g., which one/ones to regulate and how to). 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.

8.2.2 Field Emission Charge

A per-unit effluent charge can be applied to field emissions. For each combination of activities, I derive the field emission charge that would produce the same amount of field emissions 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 the same as that under the field emission limit.

8.3 Downstream Emission Control

8.3.1 Downstream Emission Limit

The total amount of downstream emissions over the 24 years horizon is 495.4 kgN/ha, or approximately 20.6 kgN/ha per year, which is set as the annual cap on downstream emissions.

The optimal activities under the downstream emission limit scenario are the same as those under the baseline scenario. As shown in Figure 8.3 , the most dramatic change in the management practices is that the operator starts to recycle the drainage water of low quality with less surface water imported. As a result, salt accumulates in the soil more quickly so the operator periodically applies a high volume of surface water to flush the salt out of the field. Similar to the FEL scenario, the operator controls the downstream emissions by taking advantage of natural processes rather than transporting liquid waste off site. However, the mechanisms under DEL and FEL are different, since the field emissions from the optimal solution under DEL (117.47 kgN/ha per year) are very high relative to that under FEL (25.2 kgN/ha per year). The onsite recycling of drainage water has two effects: (1) crops reuse some of the nitrate so that the amount of nitrate that deep percolation carries into the deep aquifer will decrease, and (2) pumping of shallow groundwater (i.e., drainage water) intensifies the process of denitrification in the saturated zone so that more nitrogen enters the atmosphere as nitrogen gas rather than enters the deep aquifer as nitrate pollution. I test the significance of each of the two effects by running the model under DEL with and without the second effect; the results suggest that the second effect is the main driving force of the overall effects introduced by recycling drainage water.

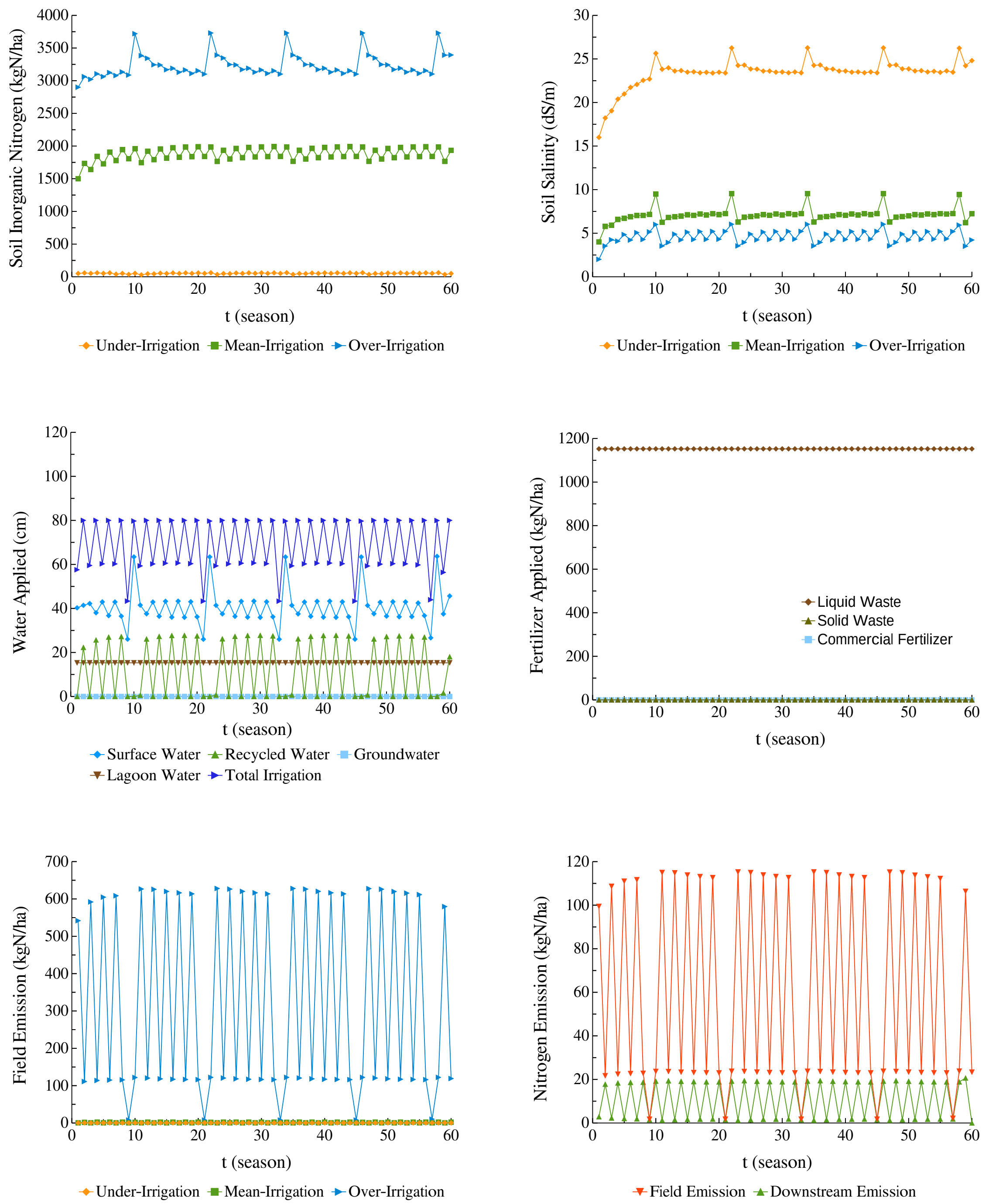

Figure 8.3: Downstream emission limit: paths of soil inorganic nitrogen, soil salinity, and field emission for each field type, paths of irrigation and fertilizer application, and seasonal nitrogen emissions under the optimal activities (flush-lagoon, $1/4$-mile furrow, corn-wheat rotation)

Compared to the optimal results of the baseline scenario, net farm income decreases by only 0.74%, with 6.73% of the crop revenue sacrificed to meet the downstream emission standard. This implies that for the case of farm nitrogen emission, quantity control of the final pollution is more cost-effective than quantity control of the intermediate pollution. The arguments for controlling inputs versus controlling intermediate pollution also applies here. The downstream emission limit creates an incentive for the operator to examine the potential benefits of recycling drainage water, which come from the natural attenuation capacity of the saturated zone.

8.3.2 Downstream Emission Charge

Similar to FEC, I derive the downstream emission charge that would produce the same amount of downstream emission as NMPs, FEL, FEC, and DEL. At an emission charge of \$2.04/kgN/ha, the operator achieves the same level of downstream emission reduction at a net income loss of 0.70% (emission charge is not counted as production cost). The optimal activity are same as and the management practices are close to those under DEL (Figure 8.4). Compared to the optimal results of DEL, the 0.04% extra savings in compliance costs are made possible by allowing occasional flexible levels of nitrate leaching over the planning horizon. This sheds lights on the advantages of emission charge over direct quantity control when seasonal or annual emissions are heterogeneous either due to inherent operation practices (e.g., different seasonal crops in rotation) or the accumulation of precursors to the pollution (e.g., accumulation of soil inorganic nitrogen).

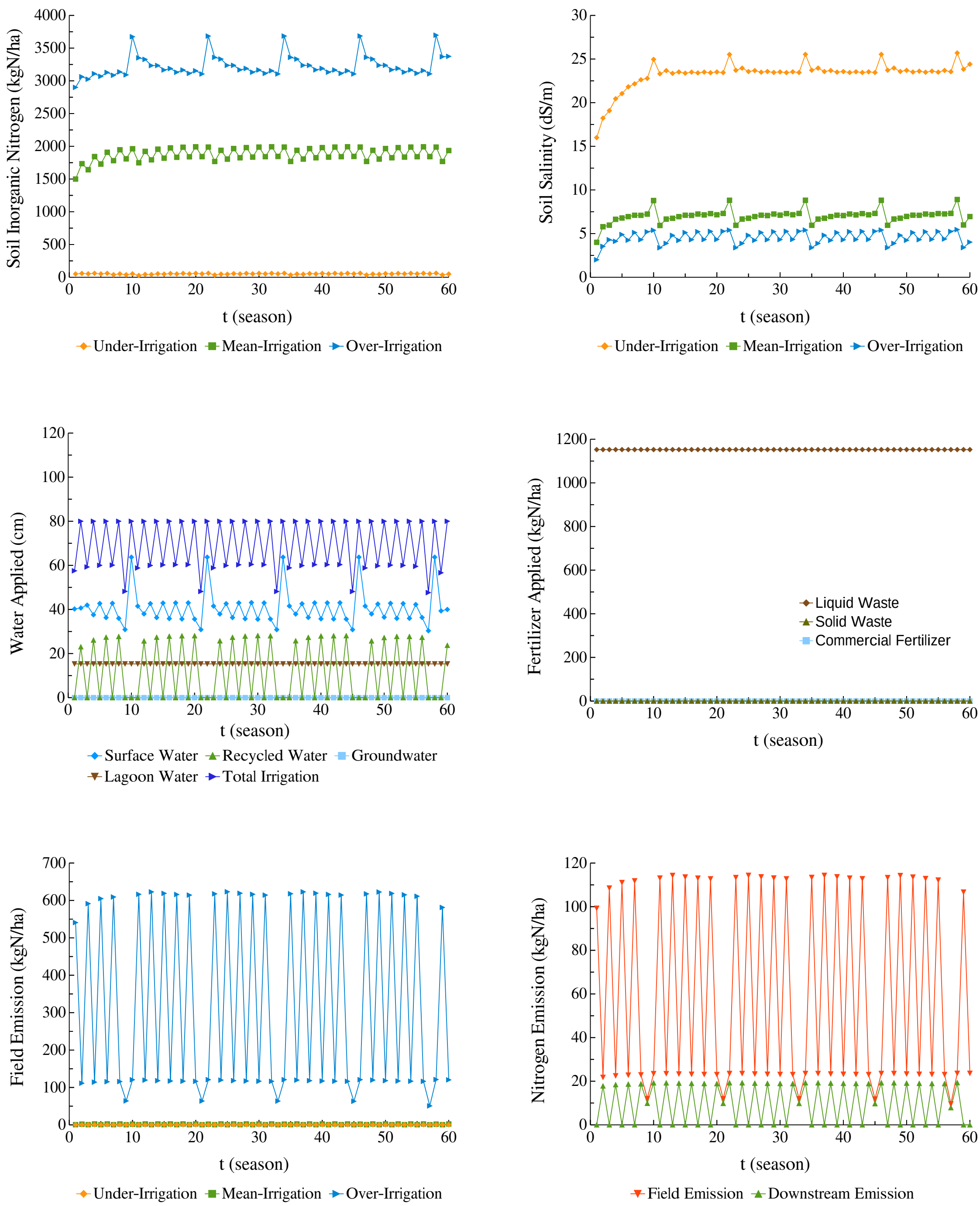

Figure 8.4: Downstream emission charge: paths of soil inorganic nitrogen, soil salinity, and field emission for each field type, paths of irrigation and fertilizer application, and seasonal nitrogen emissions under the optimal activities (flush-lagoon, $1/4$-mile furrow, corn-wheat rotation)

Chapter 9

Results and Discussions

9.1 Policy Simulation Results

Table 9.1 summarizes the policy simulation results under four sets of combined activities. For each policy, the optimal combination of activities is the one with the lowest loss of net farm income, which then is the loss of net farm income under that policy. The optimal combination of activities for NMPs, FEL, FEC, DEL, and DEC are respectively A3, A2, A2, A1, and A1, and the losses of net farm income are respectively 27.40%, 0.79%,

Table 9.1: Loss of total net farm income under alternative policy scenarios

	A1	A2	A3	A4*
NMPs	34.90%	35.00%	27.40%	27.83%
FEL	1.99%	0.79%	8.88%	9.45%
FEC†	3.42%	1.61%	9.00%	9.54%
FEC	0.72%	0.79%	8.88%	9.45%
DEL	0.74%	0.79%	8.88%	9.45%
DEC†	1.00%	1.27%	9.00%	9.54%
DEC	0.70%	0.79%	8.88%	9.45%

* A1: flush-lagoon, 1/4-mile furrow; A2: flush-lagoon, linear move; A3: scrape-tank, 1/4-mile furrow; A4: scrape-tank, linear move.
† No lump-sum return of emission charge

0.79%, 0.74%, and 0.70%. The policies targeting downstream emission are slightly more cost-effective than the policies targeting field emission, but all of them are much more cost-effetive than NMPs which targets nitrogen input.

Incentive-based mechanisms are slightly more cost-effective than quantity controls. The field emission charge is an interesting case. From the regulator's perspective, the most cost-effective outcome under FEC is a net income loss of 0.72%, which is lower than that under FEL. However, this outcome is attainable only when there is a single farm operation or when all the operations are homogeneous given the following sequences of action: the operators move first by adopting a combination of activities, then the regulator sets the charge rate accordingly, and then the charge revenue is lump-sum returned. Amacher and Malik [1998] deveop a theoretical model to compare outcomes with an emission tax and an emission standard in a cooperative bargaining framework. One of the implications from their model is that the regulator might not be able to achieve the first-best outcome if it moves first. When firms face discrete technologies, emission tax payments can create a divergence between the firms's technology preferences and that of the regulator [Amacher and Malik, 1998]. My empirical results back up their theory.

According to the results in Table 9.1, the activity A4 is dominated by A3. Also, the optimal results of quantity control and emission charge, either for field emission or for downstream emission, are very close. Therefore, for sensitivity analysis in next section, I only include NMPs, FEL, and DEL under A1, A2, and A3.

9.2 Sensitivity Analysis

9.2.1 Willingness to Accept Manure

Table 9.2 presents the results of sensitivity analysis on WTAM of surrounding land owners[1]. As they are less willing to accept manure from CAFOs, the costs for the dairy operation to meet emission targets under all the policy scenarios increase. However, the cost of complying with NMPs is most sensitive to WTAM. When WTAM decreases from 60% to 20%, the cost of NMPs increases from 27.4% to 32.74% while that of DEL from 0.74% only to 0.79%. The reason is obvious given the discussions in Chapter 8: under FEL or DEL, the operator controls nitrogen emission by taking advantage of natural processes rather than transporting half of the liquid waste off site, the cost of which largely depends on WTAM. With lower WTAM, FEL and DEL are more favorable to NMPs.

Table 9.2: Sensitivity analysis on Willingness to Accept Manure (WTAM)

WTAM	Policy	NPV loss (%)			Field Emission	Downstream Emission
		A1	A2	A3	(kgN/ha/yr)	(kgN/ha/yr)
100%	Baseline	0.00%			158.31	129.81
	NMPs	32.16%	32.26%	25.79%	25.59	20.99
	FEL	1.30%	0.78%	8.88%	25.59	20.99
	DEL	0.73%	0.78%	8.88%	134.45	20.99
60%	Baseline	0.00%			158.31	129.81
	NMPs	34.90%	35.00%	27.40%	25.17	20.64
	FEL	1.99%	0.79%	8.88%	25.17	20.64
	DEL	0.74%	0.79%	8.88%	117.47	20.64
20%	Baseline	0.00%			158.76	130.18
	NMPs	38.59%	38.69%	32.74%	23.58	19.34
	FEL	5.14%	0.83%	8.89%	23.58	19.34
	DEL	0.79%	0.83%	8.89%	134.65	19.34

[1] If a parameter changes, the initial values for the baseline scenario and its steady states usually also change. Therefore, I report relative values rather than absolute ones.

9.2.2 Denitrification Rate

The two main natural processes that the operator takes advantage of are denitrification in the unsaturated and saturated zone. Table 9.3 reports the results of sensitivity analysis on the denitrification rate in the unsaturated zone. Under the baseline scenario, the nitrogen emissions significantly decrease as the denitrification rate increases, suggesting that denitrification in the unsaturated zone is an important process. Also, as the denitrification rate increases, the decrease in nitrogen emissions under NMPs brings stricter emission regulations and thus higher net income losses. Compared to NMPs, the costs of complying with FEL and DEL are more sensitive to the denitrification rate in the unsaturated zone, since denitrification only matters for nitrogen that is already applied to the field.

Table 9.3: Sensitivity analysis on the denitrification rate in the unsaturated zone (DR-UZ)

DR-UZ	Policy	NPV loss (%)			Field Emission	Downstream Emission
		A1	A2	A3	(kgN/ha/yr)	(kgN/ha/yr)
	Baseline	0.00%			358.74	294.17
0.125	NMPs	34.71%	34.91%	27.31%	43.49	35.66
	FEL	5.89%	0.70%	8.79%	43.49	35.66
	DEL	0.62%	0.70%	8.79%	198.64	35.66
	Baseline	0.00%			158.31	129.81
0.25	NMPs	34.90%	35.00%	27.40%	25.17	20.64
	FEL	1.99%	0.79%	8.88%	25.17	20.64
	DEL	0.74%	0.79%	8.88%	117.47	20.64
	Baseline	0.00%			91.70	75.19
0.375	NMPs	35.12%	35.04%	27.44%	17.74	14.55
	FEL	0.82%	0.83%	8.93%	17.74	14.55
	DEL	0.79%	0.83%	8.93%	84.62	14.55

A change of denitrification rate in the saturated zone will affect the results under DEL but not those under FEL. Therefore, the conclusion that FEL (and DEL) is much more cost-effective than NMPs will hold for different levels of denitrification rate in the saturated zone[2].

9.3 Conclusions

This thesis uses an integrated farm level 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. Optimal decision rules from the optimization problem demonstrate best management practices, some of which are overlooked in previous studies [Ribaudo et al., 2003, Huang et al., 2005, Baerenklau et al., 2008], for CAFOs to improve their economic and environmental performance. 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 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

[2] It would be desirable to run sensitivity analysis on some other parameters such as surface water price, crop price, and milk price. Or stochastic characteristics of these prices can be investigated in future research.

that direct quantity restrictions of emissions or incentive-based emission policies are much more cost-effective than the standard approach of limiting the amount of animal waste that may be applied to the field, because policies targeting intermediate pollution and final pollution create incentives for the operator to examine the effects of other management practices to reduce pollution in addition to controlling the polluting inputs. Incentive-based mechanisms are slightly more cost-effective than quantity controls over the planning horizon, since the former gives the operator more flexibilities when seasonal or annual emissions fluctuate either due to inherent operation practices or the accumulation of precursors to the pollution. Consistent with previous studies [Schwabe, 2001], my approach demonstrates that ecosystem services can play an important role in pollution control and thus deserve more attention when designing policies. The fact that the operator can take advantage of the natural attenuation capacity of the unsaturated and the saturated zone in this study represents a good example.

Some caveats should be taken into account when interpreting the results from my model. One of the modeling assumptions is that liquid animal waste shares the same distribution system as irrigation water under the flush-lagoon system. However, in practice, liquid animal waste is less uniformly applied than irrigation water. It will be desirable to model the non-uniform distributions of both animal waste and irrigation. The results here are based on a deterministic optimization model. A future extension with stochastic components incorporated could provide a better assessment. There are consequences for the long-run between emission limits and emission charges [Spulber, 1985], which also need further study.

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Appendix A

Notation

$t = 1, \ldots, T,$ time period. The time step is G Julian years.

$g = 1, \ldots, G,$ years in a time period. G depends on crop rotation duration.

$k = 1, \ldots, K,$ seasons in a time period. 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 $1/4$-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.

$R \in \{R_1, \ldots, R_r\},$ crop rotation.

Appendix B

Hydrus Utilization

Hydrus-1D can be freely downloaded from www.pc-progress.com. In the current version, the module "Active Solute Uptake" is disabled if more than one solute is specified. Complete the following steps (From Dr. Simunek) to modify Hydrus-1D so that "Active Solute Uptake" responds to a certain solute (e.g. solute #1) and can be enabled under the case of multiple solutes.

1. Replace the h1d_calc.exe file in the folder where HYDRUS files are installed and then run it from GUI. Or run it from elsewhere using a batch file (instructions on how to do that with the Level_01.dir file are in FAQ at the HYDRUS site.
2. Run salinity as solute 1 and nitrogen as solute 2. GUI still does not support that, but the computational module does. Manipulate the Selector.in file manually.
3. If you have time variable BC, you need to specify daily potential nutrient requirement (Spot) values. This goes into cBot-2 column.
4. cRoot values for passive uptake: This should be zero for salinity and large value for nitrogen (to maximize the passive uptake).
5. Active solute uptake variables are enabled in GUI when you have two solutes. You need to add at the end of the selector.in file (note that whenever you change anything in GUI, this will be again overwritten) two lines as follows (these are parameters for active uptake. Spot is ignored since it is read from cBot-2 daily.

OmegaS	SPot	KM	cMin	OmegaW
1.0	1	0.1	0.0	f

6. Make sure that the variable lDummy is True (line 12). This initiates active uptake.

lSnow	lHP1	lMeteo	lVapor	lDummy	lFluxes	lDummy	lDummy
f	f	f	f	t	f	f	f

Appendix C

Crop Response Functions

Here I report the crop response functions with crop available water, available nitrogen, and exposed salinity as three input factors for corn,cotton, and small grains.

The crop uptake and relative yield functions are defined as

$$\Psi_e^c = \left(1 - \exp\left(-b_w^1\left(\varphi w - b_w^0\right)\right)\right)\left(1 - \exp\left(-b_n^1\left(n - b_n^0\right)\right)\right), \forall e \in \{wup, nup, ry\}$$

where

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

and 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:

$$\gamma_i = f_i^{ry}\left(s\right)$$

Similarly, the nitrate leaching function is defined as

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

with parameters $\boldsymbol{\vartheta} \equiv \left\{\vartheta_w^1, \vartheta_w^0, \vartheta_n\right\}$ depending on salinity levels:

$$\vartheta_i = f_i^{nl}\left(s\right)$$

Table C.1: Corn response function parameters

		$f_i(s)$						Ψ^c
		1	s	s^2	s^3	$\sqrt{s}$	R^2	R^2
wup	b_w^1	0.018524917	-0.000789643	1.96725E-05	-1.51849E-07	–	0.9977	0.9973
	b_w^0	-10.09159346	-0.290858616	-0.023890955	0.000749208	–	0.9816	
	b_n^1	–	–	–	–	–	–	
	b_n^0	–	–	–	–	–	–	
	d_0	-2.31659422	0.070107279	0.001985563	-0.000153755	–	0.8090	
	d_1	0.064861651	-0.000301597	-1.46937E-06	4.67029E-06	–	0.9855	
	d_2	0.290408999	0.006722907	-0.0014748	2.56167E-05	–	0.9990	
nup	b_w^1	0.041659821	-0.000624352	0.000196842	-2.65171E-06	–	0.9989	0.9956
	b_w^0	-38.4159262	0.384793384	0.060444629	-0.001488117	–	0.9992	
	b_n^1	0.007957721	1.64362E-05	-2.64997E-06	4.6767E-08	–	0.9961	
	b_n^0	-1.184947657	-0.007245857	0.001403755	-3.19919E-05	–	0.8800	
	d_0	-1.065598703	-0.028598019	0.005495742	-0.000115873	–	0.9941	
	d_1	0.05234656	0.000732666	-4.7909E-05	1.32547E-06	–	0.9778	
	d_2	0.002855162	0.000605857	-0.000114209	2.91392E-06	–	0.7705	
ry	b_w^1	0.018143115	-0.000707874	1.47853E-05	-6.51148E-08	–	0.9973	0.9973
	b_w^0	-10.31614538	-0.240814603	-0.026960413	0.000804812	–	0.9815	
	b_n^1	0.093958551	0.215302413	-0.015375301	0.000293923	–	0.7995	
	b_n^0	–	–	–	–	–	–	
	d_0	-2.170833492	0.039813245	0.003750585	-0.000184335	–	0.7736	
	d_1	0.062725976	0.000134018	-2.62252E-05	5.08705E-06	–	0.9846	
	d_2	0.298695498	0.004973273	-0.00137163	2.381E-05	–	0.9993	
nl	ϑ_w^1	0.414651136	-0.023225526	0.000408418	–	–	0.9067	0.9734
	ϑ_w^0	88.13923159	0.004554939	-0.008740355	–	–	0.9734	
	ϑ_n	0.08878923	0.006640596	-6.8241E-05	–	–	0.9878	

Table C.2: Cotton response function parameters*

		$f_i(s)$						Ψ^c
		1	s	s^2	s^3	$\sqrt{s}$	R^2	R^2
wup	b_w^1	0.025625927	-0.015073101	–	–	0.109594152	0.9983	0.8835
	b_w^0	2.881880448	-0.020488104	–	–	-0.369397872	0.9672	
	b_n^1	–	–	–	–	–	–	
	b_n^0	–	–	–	–	–	–	
	d_0	-0.199348185	-0.244594085	–	–	1.756989507	0.9997	
	d_1	0.030685925	0.003583966	–	–	-0.015752839	0.9932	
	d_2	0.469486895	0.004971243	–	–	-0.099130962	0.9749	
nup	b_w^1	0.041346997	-0.000957879	0.000132482	-2.00944E-06	–	0.7922	0.9301
	b_w^0	-81.34168775	35.93099682	-1.129046186	0.008434626	–	0.4218	
	b_n^1	0.010013821	8.44799E-05	-1.51659E-05	2.55119E-07	–	0.9233	
	b_n^0	-0.057775631	-0.040056942	-0.045260726	0.000942176	–	0.8831	
	d_0	-2.772276174	0.708780628	-0.025690687	0.000236486	–	0.3786	
	d_1	0.023926697	0.014263124	-0.000672008	7.71651E-06	–	0.3584	
	d_2	3.103187249	-2.224740853	0.148163111	-0.001977531	–	0.5799	
ry	b_w^1	0.034583661	0.049213525	-0.001273891	6.702E-06	–	–	0.7020
	b_w^0	4.51972892	-1.397983664	0.098234258	–	–	–	
	b_n^1	0.033318769	-0.002418865	0.000307345	-4.22482E-06	–	–	
	b_n^0	0.0410485	-0.003189451	5.95831E-05	-3.01448E-07	–	–	
	d_0	0.241472747	0.455173526	-0.016830118	0.000158142	–	–	
	d_1	0.028166739	-0.001251688	6.96784E-05	-4.29574E-07	–	–	
	d_2	0.404866015	-0.101796477	0.00640773	–	–	–	
nl	ϑ_w^1	0.121025853	-0.00168587	8.75232E-06	–	–	0.9285	0.9932
	ϑ_w^0	107.6679418	-0.135331412	-0.005551617	–	–	0.9966	
	ϑ_n	0.24911347	0.003728743	-1.11993E-05	–	–	0.9510	

* $\varphi = \frac{\exp(d_1 w + d_0)}{4(1+\exp(d_1 w + d_0))^2} + (d_2)^2$ for wup and nup

Table C.3: Small Grains response function parameters

		$f_i(s)$					Ψ^c
		1	s	s^2	s^3	$\sqrt{s}$	R^2
wup	b_w^1	0.020074539	0.000137967	–	–	-0.002184482	0.9952
	b_w^0	-8.761273445	0.25447724	–	–	-2.690034679	
	b_n^1	–	–	–	–	–	
	b_n^0	–	–	–	–	–	
	d_0	-4.243305301	-0.027900159	–	–	0.559798082	
	d_1	0.103594168	0.002107701	–	–	-0.015931315	
	d_2	1.078630651	-0.012689621	–	–	-0.008977404	
nup	b_w^1	0.023093471	0.001001966	-5.43379E-05	8.22543E-07	–	0.9760
	b_w^0	-27.37342722	0.302800052	0.024282858	-0.000282008	–	
	b_n^1	0.007562983	-0.000140653	2.66284E-06	-1.15131E-08	–	
	b_n^0	3.525523754	-0.960643058	0.022256413	-0.000108431	–	
	d_0	3.48195108	-5.93835677	0.2447966	-0.002065257	–	
	d_1	-0.078489412	0.008869897	-0.004225349	4.85334E-05	–	
	d_2	0.693588561	0.08834668	0.00080244	-2.26548E-05	–	
ry	b_w^1	0.021014714	-0.000379792	3.17363E-06	–	–	0.9920
	b_w^0	-9.759026759	-0.242214975	0.002548885	–	–	
	b_n^1	0.017967226	-0.000804713	6.15401E-05	–	–	
	b_n^0	0.069662204	-0.002618718	2.11647E-05	–	–	
	d_0	-3.362349424	0.087669639	-0.000767239	–	–	
	d_1	0.088470062	-0.001463417	2.37771E-05	–	–	
	d_2	0.948258212	-0.014022963	2.22753E-05	–	–	
nl	ϑ_w^1	0.721780259	0.008070633	–	–	-0.127052015	0.9732
	ϑ_w^0	81.20226075	-0.280476965	–	–	0.672129764	
	ϑ_n	0.028377048	0.001815196	–	–	-0.005647219	

Appendix D

Farm Model Parameters

Table D.1: Animal Model Parameters

Symbol	Description	Value	Source
$\mathbf{f}$	alfalfa, wheat silage, corn grain, soybean meal, and protein mix consumed by calves, heifers, and cows [kg/animal/yr]	$\begin{bmatrix} 270 & 690 & 861 \\ 861 & 2143 & 2621 \\ 522 & 102 & 3296 \\ 0 & 0 & 13 \\ 0 & 0 & 151 \end{bmatrix}$	Rotz et al. [1999] Baerenklau et al. [2008]
$\mathbf{f_1^{sw}}$	water consumption under flush-lagoon system [m^3/animal/yr]	$[1.42, 3.37, 241.77]$	Baerenklau et al. [2008] Bray et al. [2011]
$\mathbf{f_2^{sw}}$	water consumption under scrape-tank system [m^3/animal/yr]	$[1.42, 3.37, 131.24]$	Baerenklau et al. [2008] Bray et al. [2011]
$\mathbf{p^{feed}}$	feed price [$/kg]	$[0.1624, 0.1432, 0.1444, 0.3008, 0.3971]$	Baerenklau et al. [2008]
	milk produced by cohort [$/animal/yr]	$[0, 0, 9642]$	Rotz et al. [1999]
p_1^M	annualized cost of flush-lagoon system [$/cow/yr]	50	Bennett et al. [1994]
p_2^M	annualized cost of scrape-tank system [$/cow/yr]	150	Bennett et al. [1994]
p^{milk}	price of milk [$/kg]	0.31	CDFA [2010]
p^{sw}	price of imported surface water [$/$m^3$]	0.0258	Vargas et al. [2003]
$\mathbf{p^{herd}}$	prices of selling cohorts [$/animal]	$[353, 838, 633]$	Baerenklau et al. [2008]
$\mathbf{p^{fixh}}$	herd production fixed cost [$/animal/yr]	$[0, 0, 1309]$	Rotz et al. [2003]
p^{sol}	price received for dried solid waste [$/kgN]	0.14	Baerenklau et al. [2008]
p^{base}	the base price for hauling manure off-site [$/ha-cm]	201.83	Fleming et al. [1998]
p^{dist}	the hauling cost per unit distance [$/ha-cm-km]	54.02	Fleming et al. [1998]

Table D.2: Crop and Hydrologic Model Parameters

Symbol	Description	Value	Source
$prec_k$	precipitation during the summer and winter growing season [cm]	$[2, 1]$	CIMIS
δ_k	mineralization rate of organic nitrogen during the summer and winter growing season	$[0.473, 0.204]$	Chang et al. [2005]
ϕ	the fraction of applied liquid waste nitrogen that volatilizes during application	0.25	Chang et al. [2005]
λ	denitrification rate in the unsaturated zone	0.25	Meisinger and Randall [1988]
ν	field capacity of the soil [cm]	30	USDA [2007b]
nc^a	nitrogen concentration of surface water, deep groundwater, and precipitation [kgN/ha-cm]	$[0.1, 1, 0.1]$	Schans [2001]
ec^a	electrical conductivity of surface water, deep groundwater, and precipitation [dS/m]	$[0.15, 1.183, 0.1]$	Schans [2001]
p^{gw}	cost of using deep groundwater [$/ha-cm]	8.84	Ruud et al. [2003]
p^{rw}	cost of using shallow groundwater [$/ha-cm]	2.58	Ruud et al. [2003]
p^{fl}	cost of chemical fertilizers [$/kgN]	0.59	Knapp and Schwabe [2008]
$\alpha^{denitri}$	nitrate loss rate due to denitrification in the saturated zone	0.02	DeSimone and Howes [1998]
α^{miscel}	nitrate loss rate due to the other transformation processes in the saturated zone	0.16	DeSimone and Howes [1998]

Appendix E

Mathematica Code

```
(*Dynamic Model of Animal-Crop Operations_Model*)
(*Jingjing Wang, 2012*)
(*Flush - lagoon system : nonuniform application of both water and manure*)
(*Codes for data input, crop response functions,
and start points generation are available upon request*)

ClearAll[h3, soilorg, soilinorg, soilec, plantnitrogen, plantec, wateruptake,
nitrogenuptake, nitrogenleaching, nmassiw, ecmassiw]

(*State variable*)
h3All:=Array[h3, {T, G}]
soilorgAll:=Array[soilorg, {T, K, J}]
soilinorgAll:=Array[soilinorg, {T, K, J}]
soilecAll:=Array[soilec, {T, K, J}]

(*Intermediate variable*)
plantnitrogenAll:=Array[plantnitrogen, {T, K, J}]
plantecAll:=Array[plantec, {T, K, J}]
wateruptakeAll:=Array[wateruptake, {T, K, J}]
nitrogenuptakeAll:=Array[nitrogenuptake, {T, K, J}]
nitrogenleachingAll:=Array[nitrogenleaching, {T, K, J}]

(***HerdManagement***)

ClearAll[θ, h, θθ, hCull, animal, πh]
θ3All:=Array[θ, {T, G}]
h[t_, g_]:={calve, heifer, 1} * h3[t, g]
```

$\theta\theta$[t_, g_]:={calve, heifer, 1} ∗ $\theta[t, g]$
hCull[t_, g_]:={(bullcalf + calfcull) ∗ h3$[t, g]$, heifercull ∗ h3$[t, g]$, cowcull ∗ h3$[t, g]$}
animal[t_, g_]:=animalunit.{calve, heifer, 1} ∗ h3$[t, g]$

(*Economic Submodel for Herd Production*)
πh[t_]:=Sum[pmilk ∗ ymilk ∗ h3$[t, g]$ + pherd.hCull$[t, g]$ − pherd.$\theta\theta[t, g]$
−consump.pconsump.$h[t, g]$ − pwaterherd ∗ consumpWater.$h[t, g]$
−pfixherd.$h[t, g]$ − pcollect ∗ h3$[t, g]$, $\{g, G\}$]
hvars:=Flatten[{h3All, θ3All}]
hcons:=Join[hcons1, hcons2, hcons3]
hcons1:=Table[herdpermit ≥ animal$[t, g] \geq 0, \{t, T\}, \{g, G\}$]
hcons2:=Table[0==h3$[t, g-1] + \theta[t, g]$ − h3$[t, g], \{g, 2, G\}, \{t, 1, T\}$]
hcons3:=Table[0==h3$[t-1, G] + \theta[t, 1]$ − h3$[t, 1], \{t, 1, T\}$]

(***ManureManagement***)

ClearAll[lonsite, sonsite, lonsitefield, sonsitefield, lvonsite, lvoffsite, solidN,
liquidorgN, liquidinorgN, liquidN, landtosearch, rbig, raverage, phauling, πwaste]
lonsiteAll:=Array[lonsite, $\{T, K\}$];
sonsiteAll:=Array[sonsite, $\{T, K\}$];
lonsitefield[t_, k_]:=(landonsite − pond)lonsite$[t, k]$;
sonsitefield[t_, k_]:=(landonsite − pond)sonsite$[t, k]$;
lvonsite[t_, k_]:=lonsite$[t, k]$ ∗ wPERn
lvoffsite[t_, g_]:=(liquidN$[t, g]$ − Sum[lonsitefield$[t, k], \{k, 2g-1, 2g\}$]) ∗ wPERn

(*manure production*)
solidN[t_, g_]:=flow1flow8((1 − flow2)urineN + fecalN(1 − flow3)).$h[t, g]$;
liquidorgN[t_, g_]:=(1 − flow1 ∗ flow8)(urineN(1 − flow2) + fecalN(1 − flow3)).$h[t, g]$;
liquidinorgN[t_, g_]:=((urineNflow2 + fecalNflow3)flow1(1 − flow5)(1 − flow6)
+(urineNflow2 + fecalNflow3)(1 − flow1)(1 − flow4)).$h[t, g]$;
liquidN[t_, g_]:=((1 − flow1 ∗ flow8)(urineN(1 − flow2) + fecalN(1 − flow3))
+(urineNflow2 + fecalNflow3)flow1(1 − flow5)(1 − flow6)
+(urineNflow2 + fecalNflow3)(1 − flow1)(1 − flow4)).$h[t, g]$;

(*offsite manure disposal*)
landtosearch[t_, g_]:=(liquidN$[t, g]$ − Sum[lonsitefield$[t, k], \{k, 2g-1, 2g\}$])/
offsiteRate/offsiteFraction/wtam
rbig[t_, g_]:=$\sqrt{\frac{\text{landtosearch}[t,g]+\text{landonsite}}{100\pi}}$;
rsmall = $\sqrt{\frac{\text{landonsite}}{100\pi}}$;
raverage[t_, g_]:=$\frac{2\left(\text{rbig}[t,g]^2+\text{rbig}[t,g]*\text{rsmall}+\text{rsmall}^2\right)}{3(\text{rbig}[t,g]+\text{rsmall})}$
phauling[t_, g_]:=(pbase + pdist ∗ raverage$[t, g]$) ∗ lvoffsite$[t, g]$

(*Economic Submodel for Waste Disposal*)

πwaste[t_]:=Sum[(psolid $*$ (solidN$[t,g]$ $-$ Sum[sonsitefield$[t,k],\{k,2g-1,2g\}]$)
$-$phauling$[t,g]$), $\{g,G\}$]
mvars:=Flatten[{h3All, θ3All, lonsiteAll, sonsiteAll}]
mcons:=Join[hcons2, hcons3, mcons1, mcons2]
mcons1:=Table[{herdpermit $\geq$ animal$[t,g]\geq 0$,
0.5solidN$[t,g]$ $-$ sonsitefield$[t,2g-1]\geq$ eps,
0.5solidN$[t,g]$ $-$ sonsitefield$[t,2g]\geq$ eps,
0.5liquidN$[t,g]$ $-$ lonsitefield$[t,2g-1]\geq$ eps,
0.5liquidN$[t,g]$ $-$ lonsitefield$[t,2g]\geq$ eps}, $\{t,T\},\{g,G\}$]
mcons2:=Table[{lonsite$[t,k]\geq 0$, sonsite$[t,k]\geq 0$}, $\{t,T\},\{k,K\}$]

(***CropManagement***)

ClearAll[sw, gw, rw, iw, iwTotal, fertilizer, precip, psw, pgw, prw, pwatercrop,
atmosdepo, πcrop, πfarm, πpolicy, rycrop,
nFieldEmission, sFieldEmission, drainage, dw, ndw, ecdw, dp]

swAll:=Array[sw, $\{T,K\}$];
gwAll:=Array[gw, $\{T,K\}$];
rwAll:=Array[rw, $\{T,K\}$];
fertilizerAll:=Array[fertilizer, $\{T,K\}$]

precip[t_, k_]:=precipAll$[[t,k]]$
atmosdepo[t_, k_]:=atmosdepoAll$[[t,k]]$

psw[t_]:=pswAll$[[t]]$
pgw[t_]:=pgwAll$[[t]]$
prw[t_]:=prwAll$[[t]]$
pwatercrop[t_, k_]:={sw$[t,k]$, gw$[t,k]$, rw$[t,k]$}.{psw$[t]$, pgw$[t]$, prw$[t]$}

(*nonuniform irrigation*)
β[j_]:=βvec$[[j]]$
prβ[j_]:=prβvec$[[j]]$

iw[t_, k_, j_]:=$\beta[j]*$ Total[{lvonsite$[t,k]$, sw$[t,k]$, gw$[t,k]$, rw$[t,k]$}] + precip$[t,k]$;

iwTotal[t_, k_]:=Total[{lvonsite$[t,k]$, sw$[t,k]$, gw$[t,k]$, rw$[t,k]$, precip$[t,k]$}]

nmassiw[t_, k_, j_]:=precip$[t,k]*$ nprecip
+Piecewise[{{$\beta[j]*$ {sw$[t,k]$, gw$[t,k]$, rw$[t,k]$}.{nsw, ngw, ndw$[t,k-1]$}, $k\geq 2$},
{$\beta[j]*$ {sw$[t,k]$, gw$[t,k]$, rw$[t,k]$}.{nsw, ngw, ndw$[t-1,K]$}, $k==1$}}]
ecmassiw[t_, k_, j_]:=precip$[t,k]*$ ecprecip
+Piecewise[{{$\beta[j]*$ {lvonsite$[t,k]$, sw$[t,k]$, gw$[t,k]$, rw$[t,k]$}.
{eclw, ecsw, ecgw, ecdw$[t,k-1]$}, $k\geq 2$},

$\{\beta[j] * \{$lvonsite$[t,k]$, sw$[t,k]$, gw$[t,k]$, rw$[t,k]\}.$
$\{$eclw, ecsw, ecgw, ecdw$[t-1,K]\}, k == 1\}\}]$

stringMapRY = (“rycrop[t_,” <> ToString[#] <> “,j_]:=ryS” <> ToString[#] <> “[iw[t,” <> ToString[#] <> “,j],plantnitrogen[t,” <> ToString[#] <> “,j],plantec[t,” <> ToString[#] <> “,j]]”)&/@Range$[K]$
Do[ToExpression[ToExpression[“stringMapRY”, TraditionalForm]$[[i]]], \{i, K\}]$

drainage[t_, k_, j_]:=iw$[t,k,j]$ − wateruptake$[t,k,j]$
dw[t_, k_]:=Sum[pr$\beta[j]$ ∗ drainage$[t,k,j], \{j,J\}]$
sl[t_, k_]:=Sum[pr$\beta[j]$ ∗ drainage$[t,k,j]$ ∗ plantec$[t,k,j], \{j,J\}]$

nFieldEmission[t_, k_]:=Sum[pr$\beta[j]$ ∗ nitrogenleaching$[t,k,j], \{j,J\}]$
ndw[t_, k_]:=nFieldEmission$[t,k]$/dw$[t,k]$

sFieldEmission[t_, k_]:=Sum[pr$\beta[j]$ ∗ drainage$[t,k,j]$ ∗ plantec$[t,k,j], \{j,J\}]$
ecdw[t_, k_]:=sFieldEmission$[t,k]$/dw$[t,k]$

dp[t_, k_]:=Piecewise[{{dw$[t,k]$ − rw$[t,k+1], K-1 \geq k \geq 1\},$
$\{$dw$[t,k]$ − rw$[t+1,1], k == K\}\}]$

(*Economic Submodel for Crop Production*)
πcrop[t_]:=(landonsite − pond)
(Sum[Sum[pcrop$[k]$ ∗ maxycrop$[k]$ ∗ (pr$\beta[j]$ ∗ rycrop$[t,k,j]), \{j,J\}]$
−pfixcrop$[k]$ − pwatercrop$[t,k]$ − pfertilizer ∗ fertilizer$[t,k], \{k,K\}]$ − p$\beta * G)$

(***HydrologicModel***)

ClearAll[nCaptureEmission, sCaptureEmission]

nCaptureEmission[t_, k_]:=Piecewise[{{dp$[t,k]$/dw$[t,k]$ ∗ nFieldEmission$[t,k]$
∗(1 − alphaDenitr ∗ (1 + rw$[t,k+1]$/dw$[t,k]$) − alphaMiscel), $K-1 \geq k \geq 1\},$
{dp$[t,k]$/dw$[t,k]$ ∗ nFieldEmission$[t,k]$
∗(1 − alphaDenitr ∗ (1 + rw$[t+1,1]$/dw$[t,k]$) − alphaMiscel), $k == K\}\}]$
sCaptureEmission[t_, k_]:=dp$[t,k]$/dw$[t,k]$ ∗ sFieldEmission$[t,k]$

(***Incentive − basedpolicycomponent***)

πpolicy[t_]:=Sum[−swTax ∗ pwaterherd ∗ consumpWater.$h[t,g]$
+collectSub ∗ pcollect ∗ h3$[t,g], \{g,G\}]$
+(landonsite − pond)(Sum[−swTax ∗ psw$[t]$ ∗ sw$[t,k]$
−ferTax ∗ pfertilizer ∗ fertilizer$[t,k]$
−lonsiteTax ∗ lonsite$[t,k]$ − sonsiteTax ∗ sonsite$[t,k]$
−pFieEmi ∗ nFieldEmission$[t,k]$

$-\text{pCapEmi} * \text{nCaptureEmission}[t, k], \{k, K\}]$
$+\beta\text{Sub} * \text{p}\beta * G)$

(***WholeFarmModel***)

$\pi\text{farm:=Sum}\left[\text{discount}^t(\pi\text{h}[t] + \pi\text{waste}[t] + \pi\text{crop}[t] + \pi\text{policy}[t]), \{t, 1, T\}\right]$
$+\text{discount}^T\text{pherd}.h[T, G]$

$\text{vars:=Flatten[Map[Flatten[\#]\&}, \{\text{Table[rw}[e, 1], \{e, T+1, T+1\}],$
$\text{h3All}, \theta\text{3All}, \text{lonsiteAll}, \text{sonsiteAll}, \text{swAll}, \text{gwAll}, \text{rwAll}, \text{fertilizerAll},$
$\text{plantnitrogenAll}, \text{plantecAll}, \text{wateruptakeAll},$
$\text{nitrogenuptakeAll}, \text{nitrogenleachingAll},$
$\text{Map[Table[soilorg}[e, \#, j], \{e, 1, T\}, \{j, J\}]\&, \text{Table}[i, \{i, 2, K\}]],$
$\text{Table[soilorg}[e, 1, j], \{e, 2, T+1\}, \{j, J\}],$
$\text{Map[Table[soilinorg}[e, \#, j], \{e, 1, T\}, \{j, J\}]\&, \text{Table}[i, \{i, 2, K\}]],$
$\text{Table[soilinorg}[e, 1, j], \{e, 2, T+1\}, \{j, J\}],$
$\text{Map[Table[soilec}[e, \#, j], \{e, 1, T\}, \{j, J\}]\&, \text{Table}[i, \{i, 2, K\}]],$
$\text{Table[soilec}[e, 1, j], \{e, 2, T+1\}, \{j, J\}]\}]]$

$\text{equa1:=}\{0\text{==}(1-\delta[k]) * (\text{soilorg}[t, k, j] + \text{sonsite}[t, k]$
$+\beta[j] * \text{lonsite}[t, k] * \text{lorgfraction}) - \text{soilorg}[t, k+1, j],$
$0\text{==}(1-\lambda[k]) * \text{plantnitrogen}[t, k, j] - \text{nitrogenuptake}[t, k, j]$
$-\text{nitrogenleaching}[t, k, j] - \text{soilinorg}[t, k+1, j],$
$0 == (\text{ecmassiw}[t, k, j] + v * \text{soilec}[t, k, j])$
$-\text{soilec}[t, k+1, j] * (v + \text{iw}[t, k, j] - \text{wateruptake}[t, k, j])\}$

$\text{equa2:=}\{0\text{==}(1-\delta[K]) * (\text{soilorg}[t-1, K, j] + \text{sonsite}[t-1, K]$
$+\beta[j] * \text{lonsite}[t-1, K] * \text{lorgfraction}) - \text{soilorg}[t, 1, j],$
$0\text{==}(1-\lambda[K]) * \text{plantnitrogen}[t-1, K, j] - \text{nitrogenuptake}[t-1, K, j]$
$-\text{nitrogenleaching}[t-1, K, j] - \text{soilinorg}[t, 1, j],$
$0\text{==}(\text{ecmassiw}[t-1, K, j] + v * \text{soilec}[t-1, K, j])$
$-\text{soilec}[t, 1, j] * (v + \text{iw}[t-1, K, j] - \text{wateruptake}[t-1, K, j])\}$

$\text{equa3:=}\{0 == \text{soilec}[t, k+1, j] - \text{plantec}[t, k, j]\}$
$\text{equa4:=}\{0 == \text{soilec}[t, 1, j] - \text{plantec}[t-1, K, j]\}$

$\text{equa5:=}\{0\text{==soilinorg}[t, k, j] + \text{atmosdepo}[t, k] + \text{fertilizer}[t, k] + \text{nmassiw}[t, k, j]$
$+(1-\phi) * \beta[j] * \text{lonsite}[t, k] * (1 - \text{lorgfraction}) + \delta[k] * (\text{soilorg}[t, k, j]$
$+\text{sonsite}[t, k] + \beta[j] * \text{lonsite}[t, k] * \text{lorgfraction}) - \text{plantnitrogen}[t, k, j]\}$

equa6:=("wateruptake[t_," <> ToString[#] <> ",j_]-WupS" <> ToString[#] <>
"[iw[t," <> ToString[#] <> ",j],plantnitrogen[t," <> ToString[#] <>
",j],plantec[t," <> ToString[#] <> ",j]]==0")&/@Range[K]
equa7; = ("nitrogenuptake[t_," <> ToString[#] <> ",j_]-NupS" <> ToString[#] <>

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"[iw[t," <> ToString[#] <> ",j],plantnitrogen[t," <> ToString[#] <>
",j],plantec[t," <> ToString[#] <> ",j]]==0")&/@Range[K]
equa8; = ("nitrogenleaching[t_," <> ToString[#] <> ",j_]-nlS" <> ToString[#] <>
"[iw[t," <> ToString[#] <> ",j],plantnitrogen[t," <> ToString[#] <>
",j],plantec[t," <> ToString[#] <> ",j]]==0")&/@Range[K]

cons11:=Table[equa1, {t, T}, {k, K − 1}, {j, J}]
cons22:=Table[equa2, {t, 2, T + 1}, {j, J}]
cons33:=Table[equa3, {t, T}, {k, K − 1}, {j, J}]
cons44:=Table[equa4, {t, 2, T + 1}, {j, J}]
cons55:=Table[equa5, {t, T}, {k, K}, {j, J}]
cons66:=Table[equa6, {t, T}, {j, J}]
cons77:=Table[equa7, {t, T}, {j, J}]
cons88:=Table[equa8, {t, T}, {j, J}]

inequa1:={herdpermit ≥ animal[t, g] ≥ eps,
0.5solidN[t, g] − sonsitefield[t, 2g − 1] ≥ eps,
0.5solidN[t, g] − sonsitefield[t, 2g] ≥ eps,
0.5liquidN[t, g] − lonsitefield[t, 2g − 1] ≥ eps,
0.5liquidN[t, g] − lonsitefield[t, 2g] ≥ eps}
inequa2:={200 ≥ soilec[t, k, j] ≥ eps}
inequa3:={6000 ≥ lonsite[t, k] ≥ eps, 6000 ≥ sonsite[t, k] ≥ eps,
wLimit[k] ≥ iwTotal[t, k], sw[t, k] ≥ eps, gw[t, k] ≥ eps, rw[t, k] ≥ eps,
400 ≥ fertilizer[t, k] ≥ eps, dp[t, k] ≥ eps}

cons1:=Table[inequa1, {t, T}, {g, G}]
cons2:=Table[inequa2, {t, T}, {k, 2, K}, {j, J}]
cons3:=Table[inequa3, {t, T}, {k, K}]
cons4:={rw[T + 1, 1] ≥ eps}
cons:=Flatten[Table[Nest[Flatten[#]&, i, 2], {i, {cons11, cons22, cons33, cons44,
cons55, cons66, cons77, cons88, cons1, cons2, cons3, cons4, hcons2, hcons3}}]]

(*NMPs*)
inequaPolicy:={1.4 ∗ NupMax[k] ≥ lonsite[t, k] ∗ (1 − lorgfraction)
+δ[k] ∗ (sonsite[t, k] + lonsite[t, k] ∗ lorgfraction)
+fertilizer[t, k] + Sum[prβ[j] ∗ nmassiw[t, k, j], {j, J}]}
consPolicy:=Table[inequaPolicy, {t, T}, {k, K}]

(*whole farm optimization*)

πfarmMax = KnitroMaximize[{πfarm, Join[cons, Flatten[consPolicy]]}, {}, startpoint,
ReturnAsRules → True, TraceProgress → True, Hessian → "SR1", MaxIterations → 8000,
KnitroOptionList → {{"xtol", 0.1^8}, {"Algorithm", 2},
{"outmode", 1}, {"outlev", 5}, {"log_knml", 1}}]//AbsoluteTiming
```