Title
How Inefficient Are Nutrient Application Limits? A Dynamic Analysis of Groundwater Nitrate Pollution from Concentrated Animal Feeding Operations

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How Inefficient Are Nutrient Application Limits? A Dynamic Analysis of Groundwater Nitrate Pollution from CAFOs

Abstract

Animal waste from Concentrated Animal Feeding Operations (CAFOs) is a significant contributor to nitrate contamination of groundwater. To evaluate the cost-effectiveness of alternative policies for controlling nitrate pollution at the field and farm level, this article implements a structural dynamic model of a representative CAFO. The model accounts for herd management, manure handling systems, crop rotations, water sources, irrigation systems, waste disposal options, and pollutant emissions. Results show that the standard approach of limiting the amount of animal waste that may be applied to fields reduces net farm income by more than 25% whereas the most cost-effective emission-based policies reduce income only marginally. This motivates greater consideration for nonpoint source pollution control policies that target estimated emissions. Furthermore price instruments are shown to slightly outperform quantity instruments under conditions that are typical of CAFOs. The results also show that adoption of alternative technologies and practices is crucial for cost-effective abatement, and demonstrate the importance of accounting for the spatial heterogeneity of both irrigation water and salinity when designing policy mechanisms for nitrate pollution control.

Key words: CAFO, animal waste, nitrate, salinity, groundwater, dynamic optimization, pollution control policy

JEL Classification code: Q53, Q58

The growing world population, together with globally converging diets, has fueled the sustained rise in demand for food of animal origin. Between 1964-66 and 1997-99, the human population roughly doubled, while the number of domestic animals tripled (FAO 2003). In the U.S., the national average stocking density for dairy operations increased from 57 to 139 head per farm from 1992 to 2009 (USDA 2010). 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). For Kern County, one of the five leading dairy counties in the state, the average number of cows in a dairy operation is up to 3190 (CDFA 2010). Higher farm incomes due to economies of scale will likely sustain the trend toward larger and more concentrated animal feeding operations (CAFOs).

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

Nitrogen and phosphorus emissions from CAFOs have received considerable attention from regulators. Either nutrient can accelerate algae production in receiving aquatic ecosystems leading to potentially large algal blooms and a variety of problems including clogged pipelines, fish kills, and reduced recreational opportunities (USEPA 2000). Furthermore, nitrate-nitrogen in groundwater is a potential threat to public health. Two medical conditions have been linked to excessive concentration of nitrate in drinking water: methemoglobinemia (blue-baby syndrome) in infants, and stomach cancer in adults (Addiscott 1996; Powlson et al. 2008). The U.S. Geological Survey reported in 2009 that of all pollutants derived from human sources, nitrate most frequently exceeded its human health benchmark (DeSimone et al. 2009). High levels of nitrates are found most frequently in aquifers underneath agricultural regions, such as the basin-fill aquifers in the Southwest and the Central Valley aquifer system in California (DeSimone et al. 2009). Nitrate contamination of groundwater is therefore the main focus of this study.

The United States Department of Agriculture (USDA) and Environmental Protection Agency (USEPA) have endeavored to control the emissions from animal feeding operations (AFOs) since the late 1990s. The early regulations addressed the vast majority (about 95%) of AFOs with voluntary programs including environmental education, locally led conservation, financial assistance, and technical assistance (USDA and USEPA 1999). In response to the increasingly severe problem of nutrient pollution, USEPA published a new rule for CAFOs in 2003. One of the key components is nutrient management plans (NMPs). Each CAFO is required to prepare and implement a site-specific NMP for animal waste applied to land (USEPA 2003). Based on this rule, the California Regional Water Quality Control Board (CRWQCB) published a General Order for dairies in 2007.[1] Starting from 2012, the land application rate of nitrogen in the Central Valley typically is limited to 1.4 times the agronomic rate of crop nitrogen removal

[1] California's General Order is unrelated to the Natural Resources Conservation Service (NRCS) 590 standards for nutrient management. The former is a mandatory production standard (similar to provisions in the European Union Nitrates Directive) while the latter is a voluntary cost sharing program. However states often incorporate aspects of the 590 standards into their responses to the EPA rule.

(CRWQCB 2007). If implemented properly, NMPs will significantly decrease nitrogen run-off and leaching. However, developing and implementing such a plan may substantially increase operating costs for producers.

To assess the cost-effectiveness of alternative policies for controlling nitrate pollution from CAFOs, this article constructs a structural dynamic environmental-economic model of a representative dairy farm. Our results show that the standard approach of limiting the amount of animal waste that may be applied to fields reduces net farm income by more than 25% whereas the most cost-effective emission-based policies reduce income only marginally, suggesting that policies based on "estimated emissions" merit greater consideration in the nonpoint source pollution discussion. Fundamentally this is because policies targeting intermediate and final pollution create incentives for the operator to examine the effects of other management practices on pollution reduction in addition to limiting the polluting inputs. Adoption of alternative technologies and management practices is crucial and can reduce compliance costs by roughly 60-90% relative to NMPs. Furthermore, price instruments are shown to have advantages over quantity instruments under conditions that are typical of concentrated animal facilities. We also demonstrate the importance of accounting for the spatial heterogeneity of both irrigation water and salinity when designing policies for reducing nitrate pollution from CAFOs.

The Integrated Farm-Level Model

The evaluation of the economic impacts for CAFOs to comply with NMPs has received significant attention in the literature. Ribaudo et al. (2003) evaluate the costs for CAFOs to meet a nutrient standard at the farm, regional, and national levels. They use a simulation model developed by Fleming et al. (1998). The model has two components: cost of transporting and spreading manure to a specific amount of cropland, and benefits from replacing commercial fertilizer with manure nutrients. Their farm-level analysis suggests a 0.5−2.0% increase in production costs for large dairies when the willing-to-accept-manure (WTAM) by surrounding crop producers is 20% (Ribaudo et al. 2003). When competition for spreadable cropland is introduced in the regional analysis, the costs increase to 40−50% of the total net returns, not including the offsetting 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 are required to 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 an *increase* in nitrogen leaching in some areas due to expanded cropland acres and changes in crop production. Huang et al. (2005) report that 6−17% 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 market rates, which may be a tenuous assumption due to the decreasing acreage for field crops in the region. Two recent studies use Geographic Information Systems to improve the modeling of spatial transportation of manure and the associated costs at the regional level (Aillery et al. 2009; Paudel et al. 2009).

Although these studies provide useful insights into the potential economic impacts for CAFOs to comply with nutrient application standards, the models neglect dynamic aspects of animal agriculture and do not consider changes in management practices other than spreading manure on additional land and changing cropping patterns. Baerenklau et al. (2008) address these gaps with a structural dynamic whole-farm model, including herd management, crop production with non-uniform irrigation, and waste disposal. Their results indicate that the profit losses due to NMPs could be much greater than previously estimated, even in the absence of regional competition for land. The authors also point out that regulating leaching rather than nitrogen application should be more cost-effective, and they suggest that improved irrigation uniformity also should be helpful for reducing compliance costs. However they do not investigate either of these proposed options in detail.

Another factor commonly neglected in the agricultural economics literature is the natural attenuation of nitrate in the sub-surface environment. Many studies have examined nitrogen transport and transformations in the unsaturated and saturated zones. DeSimone and Howes (1998) provide a good review. For agricultural effluent, the forms and concentrations of nitrogen that reach the groundwater aquifer depend on local conditions governing physical and microbial processes and are thus highly site-specific. Singleton et al. (2007) find that saturated zone denitrification can mitigate the impact of nitrate loading at dairy operations, especially when local pumping of shallow groundwater intensifies the supply of carbon from lagoon seepage and thus increases the likelihood and rate of denitrification. Despite these findings, nitrate attenuation is either modeled as exogenous to farm management or is missing from previous economic analyses of groundwater nitrate pollution. The present study remedies this by linking the attenuation rate to farm management decisions.

Similar to Baerenklau et al. (2008), this article constructs a structural dynamic environmental-economic model to investigate cost-effective policies for nitrate pollution control at the farm level. The model departs from and builds upon previous work in several important ways. First, we incorporate a set of newly-developed crop response functions that account for the effects on crop yield and nitrate leaching of available water, available nitrogen, and soil salinity. Such three-input response functions have not been used previously in studies of agricultural nonpoint source pollution control. Second, we explicitly consider soil salinity dynamics and the role it plays in irrigation decisions and thus nitrate leaching rates. Third, we allow for a much richer set of control variables that includes alternative manure handling systems, irrigation systems, crop rotations, and irrigation water sources with varying quality. Fourth, we model subsurface hydrogeochemical processes that not only help to determine downstream nitrate loading rates, but also can be affected by some of the control variables in potentially advantageous ways (e.g., intensified nitrate attenuation). And fifth, we use the model to simulate the effects of alternative pollution control polices, including price and quantity instruments levied on estimated emissions.

Our integrated farm-level model is adapted from Baerenklau et al. (2008). Figure 1 summarizes the key inputs and outputs (bold text), choice variables (ovals), and sub-components (shaded). Although the model is calibrated to a representative dairy farm in California's San Joaquin Valley, it can be adapted to other types of AFOs. The three main components that make up the full model are animal, crop, and hydrologic models. The animal model and the crop model have corresponding economic submodels, which together constitute the whole farm economic model. The hydrologic model simulates the pollutant emissions both at the field level and at the farm level. Complete details of each of these model components are provided in a technical appendix (Wang and Baerenklau 2014a); below we provide a general overview.

The objective of the model farmer is to maximize net farm income over the designated planning horizon. Net farm income equals the net incomes from herd production, waste management, and crop production less the environmental policy costs. The objective function is

$$\max\left[\sum_{t=1}^{T}\eta^{t}\pi_{t}+\eta^{T}S^{\top}\right]=\max\left[\sum_{t=1}^{T}\eta^{t}\left(\pi_{t}^{herd}+\pi_{t}^{waste}+\pi_{t}^{crop}-\pi_{t}^{policy}\right)+\eta^{T}S^{\top}\right] \qquad (1)$$

where η is the discount rate. The net income from herd production (π_t^{herd}) is equal to the revenue from milk and meat production less the total production cost. These revenues and costs are largely determined by the herd size which, along with certain farm and waste management practices, also determines the amount and content of waste produced. An important waste management decision that we explicitly model is the type of manure handling system used on the farm. Two common manure handling systems are considered in this study: flush-lagoon and scrape-tank. The scrape-tank system is more labor and capital intensive but uses 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 irrigation systems. Therefore, 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. Currently the flush-lagoon system is used in about two-thirds of all California dairies and typically employed in the Central Valley (Hurley et al. 2007). For each system, the waste management component of the objective function (π_t^{waste}) accounts for the revenues that can be earned from selling dried solid waste and the cost of transporting excess liquid waste off-site.

The net income from crop production (π_t^{crop}) equals gross returns less both fixed and variable costs. The fixed production costs include operating costs such as seed, herbicide, labor, and machinery but not overhead costs. The variable costs include irrigation and fertilizer costs. Improved irrigation uniformity has been shown to be a promising method of cost reduction under environmental regulations. Therefore we explicitly consider irrigation system choice and the effects of system non-uniformity on crop production and nitrate leaching. We also account for the combined effects of water, nitrogen, and salinity on crop growth and leaching using functions developed by Wang and Baerenklau (2014b).

The remaining terms in the objective function account for incentive-based policy costs (π_t^{policy}), when applicable, and the salvage value of the herd ($\eta^T S^\top$). The objective function is maximized subject to transition equations, mass balance requirements, non-negativity constraints, herd permit limits, and command-and-control policy constraints, when applicable. The model has a total of ten state variables: one for herd size and nine describing three soil characteristics (soil organic nitrogen, soil inorganic nitrogen, and soil salinity) across three types of field subareas. As described in the appendix, we use these types to account for irrigation system non-uniformity by modeling the cropped area as though it is comprised of three distinct sub-areas: one area receives less than the average amount of applied water (under-irrigated subfield), one receives the average amount (mean-irrigated subfield), and one receives more than the average (over-irrigated subfield).

Currently the stochastic nature of parameters such as milk and crop prices are not included in the model, so rather than set up the problem in a dynamic programming framework, we treat it 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*. For simplicity and tractability, we assume 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. We are interested in two manure handling systems (flush-lagoon and scrape-tank), two irrigation systems (1/4-mile furrow and linear move system), and six patterns of crop rotation (corn or alfalfa for summer; wheat, oat, or triticale for winter), resulting in a total of 24 alternative combinations of discrete management practices (hereafter referred to as alternative activity sets A1,…,A24). For example, the activity set A1 would be flush-lagoon, 1/4-mile furrow, and corn-wheat rotation. Conditional on an activity set, the operator determines the optimal number of cows to buy or sell, and the optimal amounts of liquid animal waste, solid animal waste, surface water, deep groundwater, shallow groundwater and commercial fertilizers to apply during each season of each year of the planning period. Solving the optimization problem similarly for each possible activity set identifies the set that maximizes net farm income (in the following sections we denote the activity set associated with the optimal solution as "the optimal activity set").

Baseline Simulation

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[2]. Therefore, the following analyses are based on the results of the first 24 years. All revenues and costs are reported in 2005 dollars. Table 1 compares the steady state values of the baseline scenario against available data for our study site (van der Schans 2001) and from other sources. 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. Animal numbers are similar to those at the study site, with differences due to off-farm rearing of calves and heifers. The herd size also remains steady through time, constrained by the herd permit. Annual income per cow is higher than the comparison source due to different assumptions about milk production per cow in different areas of the country. The field emission of nitrogen is low compared to the study site, which is

[2] Here we use the term "steady state" in a broader sense than just a terminology associated with infinite deterministic dynamic programming problems. Many of our variables exhibit regular fluctuations in their steady states.

probably because we assume a deeper root zone of 3 meters.[3] In summary, our model appears to be calibrated well.

It is also worth noting that the operator does not apply commercial fertilizer or solid waste on site in the baseline scenario. In reality, farmers are usually concerned about certain risks associated with manure fertilizer, such as pathogens and weeds and the fact that organic nitrogen is not immediately plant-available. Therefore they also use some commercial fertilizer. This is why we see the difference between our simulated value and the comparison value for applied fertilizer in Table 1. We do not consider these issues for the dairy operator but for surrounding land owners we use three levels of WTAM (20%, 60%, and 100%) to account for these concerns and perform sensitivity analysis. The policy simulations presented below are for the WTAM level of 60%. Figures 2(a)-2(c) display the optimal paths of soil organic nitrogen, soil inorganic nitrogen, and soil salinity, which vary depending on the field type. The level of soil organic nitrogen is highest in the over-irrigated subfield and lowest in the under-irrigated subfield, while the level of soil salinity is lowest in the over-irrigated subfield and highest in the under-irrigated subfield. 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 subfield that receives more animal waste, and more salts accumulate in the subfield that receives less irrigation water. The optimal decision rule for seasonal irrigation, as shown in figure 2(d), suggests that in order to maintain a

[3] Here and elsewhere, we distinguish between "field emissions" (vertical leaching to underlying groundwater) and "downstream emissions" (lateral migration to off-site ecosystems) for purposes of subsequent policy simulations. Refer to the appendix for more details.

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

Figures 2(e) and 2(f) show the field and downstream emissions of nitrogen, by field type and overall. The figures indicate that flushing of salts also carries more nitrates from the root zone to groundwater. The effect is only and especially significant for the mean-irrigated subfield. For the under-irrigated subfield, there is no excess water even during flushing. For the over-irrigated subfield, there is enough excess water to carry all leachable nitrates through the soil even if there is no flushing. The overall effect is clear in figure 2(f), and has potential downstream implications, as well.

Table 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 contributor to field emissions is the over-irrigated subfield, due to the high non-uniformity of the 1/4-mile furrow system. The over-irrigated field type makes up only 18% of the field, receives only 31% of total irrigation, produces only 20% of total crop yield, but accounts for 77% of total field emission of nitrogen.

To further illustrate the effects of non-uniform irrigation, we report similar information in table 3 from the optimization results under an alternative activity set where the linear move system is adopted instead of furrow. Compared to the baseline scenario, the amount of applied water decreases by 6%, but the total relative yield increases by 3% and the total amount of nitrogen field emission decreases by 46%. With the linear move irrigation system, the operator no longer periodically applies 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 acceptable levels 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 95% decrease. Also, nitrate leaching from the over-irrigated subfield decreases by 31% because of the improved uniformity of water and waste distribution (refer to Appendix figure A2.1). However the net farm income is lower under this activity set due to the higher cost of the linear move system. This implies that a relatively simple policy of subsidizing more uniform irrigation systems might be able to achieve a substantial reduction in field emissions, which suggests that elements of cost-sharing and technical assistance should be part of the policy discussion.

A switch from the flush-lagoon system to the scrape-tank system can also effectively reduce nitrate leaching, but through different mechanisms (refer to Appendix figure A2.2). As previously discussed, animal waste collected through the scrape-tank system is uniformly spread over the field using tractors while irrigation water is not. Therefore, under this alternative activity set, the over-irrigated subfield has the smallest amount of both nitrogen and salts in soil because it has the highest level of leaching. For the same reasons, the steady state level of soil salinity for the over-irrigated subfield is lower than that under the baseline scenario. Similarly, the steady state level of soil salinity for the under-irrigated subfield is higher than that under the baseline scenario. As shown in table 4, compared to the baseline scenario, the amount of nitrate leaching from the over-irrigated subfield significantly decreases under this alternative waste management activity. The mean-irrigated subfield 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. For the whole field, the amount of applied water, the total relative yield, and the total field emissions decrease by 2%, 0.5% and 33% respectively, compared to the baseline scenario. 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 emissions.

Policy Simulations

We simulate five alternative policy scenarios: nutrient management plan (NMP), field emission limit (FEL), field emission charge (FEC), downstream emission limit (DEL), and downstream emission charge (DEC). NMP is a quantity restriction on a polluting input. FEL and FEC are, respectively, a quantity restriction and an emission charge on the intermediate pollution (i.e., nitrates leached to groundwater that has not yet migrated off-site), while DEL and DEC are, respectively, a quantity restriction and an emission charge on the final pollution (i.e., nitrates in groundwater that migrates off-site and enters the ecosystem).

The policy simulations are designed to quantify the inefficiency of NMP compared to emission-based policies and to investigate the relative cost-effectiveness of price versus quantity mechanisms. Therefore we use the field and downstream emissions from the optimal activity set under NMP as reference points, and adjust the emission limits and charges under the FEL and FEC so that they achieve the same level of field emissions (and downstream emissions) as under NMP. Similarly, we adjust the emission limits and charges under the DEL and DEC to achieve the same downstream level of nitrogen loading as under NMP. Reductions in net farm income are then compared. For each policy simulation, we assume the operation is initially at the steady state derived from the baseline scenario and then solve for the dynamically optimal practices under each policy.

Nutrient Management Plan

NMPs are included in the model as constraints on seasonal applications of inorganic nitrogen. We construct the constraints to be consistent with the NMP requirement in the published General Order: the sum of nitrogen from applied animal waste (inorganic nitrogen plus the amount of organic nitrogen that is mineralized during that season), fertilizers, irrigation water, and atmospheric deposition must be no greater than 1.4 times the agronomic uptake rate (CRWQCB 2007).

Under the NMP scenario, the operator optimally selects scrape-tank as the manure handling system, 1/4-mile furrow as the irrigation system, and corn-wheat as the crop rotation under the NMP scenario. Due to the NMP constraint, the operator hauls almost half of the liquid waste off site. As shown in figures 3(a)-3(c), this results in significant decreases in soil organic nitrogen and soil inorganic nitrogen, and moderate decreases in soil salinity, for the mean- and over-irrigated subfields compared to the baseline scenario in figure 2. Another change in the management practices is the irrigation pattern (figure 3(d)). Although the total amount of surface water applied over the planning horizon increases little, the water is smoothly applied without flushing. This is why the soil salinity of the under-irrigated subfield remains high. Compared to the baseline scenario, both the field emission and the downstream emission of nitrogen (figures 3(e) and 3(f)) decrease by 84.1% under NMP. Total crop revenue increases by 6.2% but the operator still suffers a heavy net income loss of 27.4%, primarily due to the high cost of offsite waste hauling.

Field Emission Limit

Under the NMP scenario, the total amount of field emissions over the 24 year horizon is 604.2 kgN/ha, or approximately 25.2 kgN/ha per year. We therefore set this as the annual field emissions limit.[4]

The optimal activities under the FEL scenario are flush-lagoon as the manure handling system, linear move as the irrigation system, and corn-wheat as the crop rotation. Figure 4 displays the main results. Unlike under the NMP 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 (figures 4(a) and 4(b)). 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 leaching nitrate and salt into the aquifer, the operator takes advantage of the natural denitrification processes to reduce field emissions.[5]

The reduction in irrigation mainly happens in summer, since the winter crop is more salt-tolerant and, under the baseline scenario, the field emission from summer cropping is five times greater than that from winter cropping. Less irrigation leads to higher levels of soil salinity in the subfields (figure 4(c)), which can reduce crop yields. Flushing is again absent from the

[4] We also test a 6-year cap of 151 kgN/ha for field emissions. The optimal solution is similar to that under the annual cap, with the net farm income slightly higher due to the added flexibility provided by this longer time horizon. In practice, emission limits or charges would be imposed on estimated (modeled) nitrogen emissions.

[5] Nitrous oxide, which is a powerful greenhouse gas, can be produced during the denitrification process and escape to the atmosphere. Soil and aquifer conditions determine how much nitrous oxide is produced. In this study we focus on how denitrification can affect water quality (i.e., how much nitrates are converted to gaseous compounds) and leave its potential effects on air quality for future research.

irrigation schedule (figure 4(d)) and, by construction, field and downstream emissions (figure 4(e) and 4(f)) are the same as those under NMP. However, compared to the baseline scenario, net farm income decreases by only 0.79%, with 7.20% of crop yields sacrificed to meet the field emission standard. This implies that quantity control of intermediate pollution is much more cost-effective than quantity control of polluting inputs for the case of nitrogen emissions.

Field Emission Charge

A per-unit effluent charge also could be applied to estimated field emissions. For each activity set, we derive the field emission charge that would produce the same amount of field emissions as the NMP and FEL scenarios. A lump-sum return of emission charges does not alter marginal conditions in our model, and thus does not affect the optimal activities for a given scenario. At an emission charge of $2.50/kgN/ha, the operator achieves the same level of emission reduction at a net income loss of 0.79%. The optimal activity set and other management practices are the same as that under the field emission limit.

Downstream Emission Limit

The total amount of downstream emissions over the 24 year horizon under both NMP and FEL is 495.4 kgN/ha, or approximately 20.6 kgN/ha per year.[6] We therefore set this as the annual limit 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 5, the paths for soil organic nitrogen, soil inorganic nitrogen, and soil salinity (figures 5(a)-5(c)) are similar to those under FEL. The most dramatic change in management practices is that the operator now recycles shallow groundwater

[6] Downstream emissions are less than field emissions due to denitrification and other transformation processes of nitrate in the saturated zone.

(low quality drainage water) and imports less surface water (figure 5(d)). 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 subsurface 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 (figures 4(e) and 4(f)) are very high relative to that under FEL (figures 5(e) and (f)). The onsite recycling of drainage water under DEL has two effects: (1) crops reuse some of the nitrate so that the amount of nitrate carried into the deep aquifer decreases, and (2) pumping of shallow groundwater intensifies the process of denitrification in the saturated zone so that more nitrogen enters the atmosphere as nitrogen gas rather than entering the deep aquifer as nitrate pollution. We 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 substantially larger.

Compared to 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. Thus quantity control of final pollution is only slightly more cost-effective than quantity control of intermediate pollution.

Downstream Emission Charge

Similar to FEC, we derive the downstream emission charge that would produce the same amount of downstream emission as the NMP, FEL, FEC, and DEL scenarios. 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% (again assuming emission charges are returned as a lump sum). The optimal activities are the same as, and the management practices are close to, those under DEL (Appendix figure A2.3). Compared to DEL, the 0.04% extra savings in compliance costs are derived from the additional flexibility in levels of nitrate leaching over the planning horizon. This demonstrates one advantage of an emission charge over an emissions limit when seasonal or annual emissions fluctuate either due to operating practices (i.e., different seasonal crops in rotation) or the cyclical accumulation of precursors to the pollution (i.e., soil inorganic nitrogen). However in this case the effect is relatively small.

Summary of Results

Table 5 summarizes the policy simulation results under four sets of activities.[7] The optimal activity sets (i.e., smallest net income loss) for NMP, FEL, FEC, DEL, and DEC are respectively A3, A2, A1, A1, and A1, and the associated losses are respectively 27.40%, 0.79%, 0.72%, 0.74%, and 0.70% of net farm income. The policies targeting downstream emissions are slightly more cost-effective than the policies targeting field emissions, and price mechanisms (emission charges) are slightly more cost-effective than quantity controls. Regardless all emission-based policies are substantially more cost-effective than NMP which targets nitrogen input.

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 type and given the following sequence 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) develop a theoretical model to compare outcomes with an emission tax and an emission standard in a cooperative bargaining framework. One of the

[7] For all the scenarios, it is always optimal to select the corn-wheat rotation. The intuition is that alfalfa obtains a high percentage of its nitroen by fixing the nitrogen gas in the atmosphere and thus takes up less nitrogen from soil compared to other crops; the three winter forages (wheat, oat, and triticale) have similar ability to take up nitrogen but wheat is optimally selected due to its relatively high price in the past decade.

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' technology preferences and that of the regulator (Amacher and Malik, 1998). Our empirical results are consistent with their theory.

Sensitivity analyses on both WTAM and the denitrification rate (Appendix tables A2.1 and A2.2) demonstrate that the results for emission-based policies are very stable in these dimensions whereas net income losses under NMP can be significantly affected by changes in WTAM. We also investigate sensitivity to farm size by running simulations with the stocking density set at 75% and 125% of the original value. Results are similar to those in the original scenarios: in both cases there are significant losses under NMPs and much smaller losses under emission-based policies. For the 75% stocking density, the NMP loss is around 15% while the emission-based losses are less than 1%. For the 125% stocking density, the NMP loss is over 30% while the emission-based losses are less than 2%. Technology choices remain unchanged from the original scenarios but there is more (less) reliance on off-site waste transport for larger (smaller) farms.

Conclusions

This article uses an integrated farm level model to empirically quantify the cost-effectiveness of alternative policy mechanisms for controlling nitrate pollution from CAFOs at the field and farm level. The optimized characteristics of the animal-crop operation without regulations are consistent with available data. Our policy simulations both confirm previous work that predicts significant income losses from regulating nutrient application rates, and also quantify the potentially large inefficiency associated with this type of input regulation relative to emission-based policies. We also show that incentive-based emission controls are slightly more cost-effective than quantity-based emission controls over the planning horizon because the former gives the operator greater flexibility when seasonal or annual emissions fluctuate either due to operating practices or the cyclical accumulation of precursors to pollution, which are typical of CAFOs. Quantity-based controls with longer time horizons have similar effects (see footnote 4) and might be implemented in practice by allowing banking (but not trading) of emission credits when emissions are below short-term quantity limits.

Given the persistent nutrient-related environmental problems that are common to intensively farmed regions of the world, the trend toward larger and more concentrated AFOs, and the potential for mandatory nutrient application limits to be adopted elsewhere, these results have clear policy relevance. Foremost, they suggest that emission-based policies should not be dismissed out-of-hand as too expensive for controlling nonpoint source pollution in practice. The presumably higher information costs associated with emission-based policies should instead be weighed against the potentially large savings in abatement costs.[8] While there may be a tendency to assume that the information costs will be excessive for nonpoint source problems, this may not be the case for policies based on estimated (or modeled) emissions such as those examined here. Models already are used to develop NMPs, establish Total Maximum Daily Loads, and set emission trading ratios for nitrogen in the U.S., and to determine critical loads for nitrogen in the European Union, so extending them to emission-based policies seems reasonable. In light of the results presented here, and given a choice between meeting a land application limit or incurring costs to provide information to facilitate emissions modeling and regulation, animal facility operators may actually prefer the latter; yet the nonpoint source pollution policy discussion is not moving in that direction.

[8] Millock et al. (2002) provide a relevant theoretical analysis of this tradeoff and Kurkalova et al. (2004) present the empirical case of carbon sequestration.

In addition, and consistent with previous studies (e.g., Schwabe, 2001), our approach demonstrates that ecosystem services (here, subsurface denitrification) can play an important role in achieving cost-effective pollution control and thus deserve more attention when designing policies. We also show that optimal technologies and management practices (e.g., waste disposal methods, irrigation system, manure handling system, irrigation pattern, and shallow groundwater recycling) vary across nitrate control policies and moreover can significantly reduce compliance costs. For all of our emission-based policies, optimal technology and management choices reduce compliance costs by roughly 60-90% relative to NMPs (Table 5). This underscores the need to incorporate such choices into policy simulation models, and also suggests that elements of cost-sharing, technical assistance, and R&D funding should be part of the policy discussion. Furthermore we demonstrate the importance of accounting for salinity effects in the debate about nitrogen. We do this by implementing crop response functions that integrate the effects of available water, available nitrogen, and exposed salinity. These relationships, combined with field-level heterogeneity and soil nitrogen transformations, are shown to significantly affect both the pattern and quantity of nitrate emissions under alternative policies. Modeling of both temporal and spatial dynamics of soil characteristics is necessary to account for these factors and thus should be incorporated in future research.

Some additional caveats to consider when interpreting our results include the following. First is the assumption that liquid animal waste shares the same distribution system as irrigation water under the flush-lagoon system. However, in practice, liquid animal waste typically is less uniformly applied than irrigation water since it likely enters the system at a single point whereas irrigation water enters at multiple points. It would be desirable to model the different non-uniform distributions of animal waste and irrigation water but we lack the information to do this here. Second, our results are based on a deterministic optimization model. A future extension with stochastic components (e.g., prices, weather) could provide an improved assessment. Third, we do not consider the potential cross-media effects of nitrate regulations. Baerenklau et al. (2008) show that these could be substantial for the case of transferring nitrate to ammonia when groundwater emissions are regulated but air emissions are not. Neither that study nor this one considers the potential costs associated with increased nitrous oxide emissions. Ideally the social costs associated with such policy-induced cross-media emissions should be incorporated into the policy analysis. Fourth, our model is based on a single representative dairy farm. A future extension would be to investigate impacts of alternative policies at the regional level. Last, we do not consider long-run industry-level effects such as entry/exit incentives that can differ across policy mechanisms. Given the ongoing migration of large dairy farms both within and beyond California's borders, this would seem to be a fruitful topic for further study.

References

Addiscott, T. 1996. Fertilizers and nitrate leaching. *Issues in Environmental Science and Technology* 5, 1–26.

Aillery, M., N. Gollehon, V. Breneman, and S. Bucholtz 2009. Modeling firm spatial interdependence using national data coverages: A regional application to manure management. *Natural Resource Modeling* 22(1), 42–66.

Amacher, G. and A. Malik 1998. Instrument choice when regulators and firms bargain. *Journal of Environmental Economics and Management* 35(3), 225–241.

Baerenklau, K., N. Nergis, and K. Schwabe 2008. Effects of nutrient restrictions on confined animal facilities: Insights from a structural-dynamic model. *Canadian Journal of Agricultural Economics* 56(2), 219–241.

CDFA 2010. California dairy statistics 2010. Technical report, California Department of Food and Agriculture (CDFA), Dairy Marketing Branch, California.

Cooley, H., J. Christian-Smith, and P. Gleick 2009. Sustaining California agriculture in an uncertain future. Technical report, Pacific Institute.

CRWQCB 2007. Waste discharge requirements general order for existing milking cow dairies. Technical Report Order No R5-2007-0035, California Regional Water Quality Control Board, Central Valley Region.

DeSimone, L., P. Hamilton, and R. Gilliom 2009. Quality of water from domestic wells in principal aquifers of the United States, 1991-2004. Technical Report Scientific Investigations Report 2008-5227, U.S. Geological Survey.

DeSimone, L. A. and B. L. Howes 1998. Nitrogen transport and transformations in a shallow aquifer receiving wastewater discharge: a mass balance approach. *Water Resources Research* 34(2), 271–285.

FAO 2003. World agriculture: towards 2015/2030. Technical report, Food and Agriculture Organization of the United Nations, Rome.

Fleming, R., B. Babcock, and E. Wang 1998. Resource or waste? the economics of swine manure storage and management. *Review of Agricultural Economics* 20(1), 96.

Huang, W., R. Magleby, and L. Christensen 2005. Economic impacts of EPA's manure application regulations on dairy farms with lagoon liquid systems in the southwest region. *Journal of Agricultural and Applied Economics* 37(1).

Hurley, S., J. Ahern, and D. Williams 2007. An analysis of the tradeoffs between policy instruments to induce dairy producers in california to participate in a centralized digester. In *Proceedings of the 2007 American Agricultural Economics Association Annual Meeting*, pp. 1–25.

Kaplan, J., R. Johansson, and M. Peters 2004. The manure hits the land: Economic and environmental implications when land application of nutrients is constrained. *American Journal of Agricultural Economics* 86(3), 688–700.

Knapp, K. and K. Schwabe 2008. Spatial dynamics of water and nitrogen management in irrigated agriculture. *American Journal of Agricultural Economics* 90(2), 524–539.

Kurkalova, L., C. Kling, and J. Zhao 2004. Multiple benefits of carbon-friendly agricultural practices: empirical assessment of conservation tillage. *Environmental Management* 33(4), 519–527.

Lubowski, R., S. Bucholtz, R. Claassen, M. Roberts, J. Cooper, A. Gueorguieva, and R. Johansson 2006. Environmental effects of agricultural land-use change: The role of economics and policy. Technical report, Economic Research Service, United States Department of Agriculture.

Paudel, K., K. Bhattarai, W. Gauthier, and L. Hall 2009. Geographic information systems (GIS) based model of dairy manure transportation and application with environmental quality consideration. *Waste Management* 29(5), 1634–1643.

Powlson, D. S., T. M. Addiscott, N. Benjamin, K. G. Cassman, T. M. de Kok, H. van Grinsven, J.-L. L'hirondel, A. A. Avery, and C. van Kessel 2008. When does nitrate become a risk for humans? *Journal of Environmental Quality* 37 (2), 291–295.

Ribaudo, M., J. Kaplan, L. Christensen, N. Gollehon, R. Johansson, V. Breneman, M. Aillery, J. Agapoff, and M. Peters 2003. Manure management for water quality costs to animal feeding operations of applying manure nutrients to land. Technical Report Agricultural Economic Report No. 824, Economic Research Service, United States Department of Agriculture.

Schwabe, K. 2001. Nonpoint source pollution, uniform control strategies, and the neuse river basin. *Review of Agricultural Economics* 23(2), 352–369.

Singleton, M. J., B. K. Esser, J. E. Moran, G. B. Hudson, W. W. McNab, and T. Harter 2007. Saturated zone denitrification: potential for natural attenuation of nitrate contamination in shallow groundwater under dairy operations. *Environmental science & technology* 41(3), 759–765.

USDA 2010. Agricultural statistics 2010. Technical report, United States Department of Agriculture (USDA), National Agricultural Statistics Service, Washington.

USDA and USEPA 1999. Unified national strategy for animal feeding operations. Technical Report EPA-833-R-99-900, United States Department of Agriculture (USDA) and the United States Environmental Protection Agency (USEPA).

USEPA 2000. National water quality inventory: 1998 report to congress. Technical Report EPA-841-R-00-001, United States Environmental Protection Ageny (USEPA), Washington, DC.

USEPA 2003. National pollutant discharge elimination system permit regulation and effluent limitation guidelines and standards for concentrated animal feeding operations (CAFOs); final rule 2003. *Federal Register* 68(29), 7176–7274.

van der Schans, M. 2001. Nitrogen leaching from irrigated dairy farms in Merced County, California. Master's thesis, Wageningen University.

Wang, J. and K.A. Baerenklau 2014a. AEPP appendix: How inefficient are nutrient application limits? A dynamic analysis of groundwater nitrate pollution from CAFOs? Unpublished manuscript. Available at: http://ageconsearch.umn.edu/.

Wang, J. and K.A. Baerenklau 2014b. Crop response functions integrating water, nitrogen, and salinity. *Agricultural Water Management* 139, 17–30.

Table 1 Model validation

Variables	Units	Steady State Value	Comparison Value	Comparison Source
Calves	# of animals	723	517	van der Schans (2001)
Heifers	# of animals	578	308	van der Schans (2001)
Milk cows	# of animals	1445	1731	van der Schans (2001)
Heifers purchased	# of animals	0	NA	
Annualized profit	$/cow	757.2	1309	Rotz et al. (2003)
Field Emission	kgN/ha-yr	158.3	202-660	van der Schans (2001)
Downstream Emission	kgN/ha-yr	129.8	NA	
Total applied water	cm/yr	150.4	124	van der Schans (2001)
Applied surface water	cm/yr	116.8	NA	
Applied groundwater	cm/yr	0	NA	
Recycled drainage water	cm/yr	0	NA	
Applied commercial fertilizer	kgN/ha-yr	0	130-280	van der Schans (2001)
Applied liquid waste	kgN/ha-yr	2304.8	NA	
Applied solid waste	kgN/ha-yr	0	NA	
Irrigation system		1/4-mile furrow	1/4- and 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)

Table 2 Baseline: aggregate irrigation, relative yield, and field emission of nitrogen for each field type under the optimal activity set (flush-lagoon, 1/4-mile furrow, corn-wheat rotation)

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

Table 3 Baseline: aggregate irrigation, relative yield, and field emission of nitrogen for each field type under an alternative activity set (flush-lagoon, linear move, corn-wheat rotation)

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

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

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

Table 5 Loss of total net farm income under alternative policy scenarios

	A1[a]	A2[a]	A3[a]	A4[a]
NMP	34.90%	35.00%	27.40%	27.83%
FEL	1.99%	0.79%	8.88%	9.45%
FEC[b]	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[b]	1.00%	1.27%	9.00%	9.54%
DEC	0.70%	0.79%	8.88%	9.45%

[a] A1: flush-lagoon, 1/4-mile furrow, corn-wheat rotation; A2: flush-lagoon, linear move, corn-wheat rotation; A3: scrape-tank, 1/4-mile furrow, corn-wheat rotation; A4: scrape-tank, linear move, corn-wheat rotation.

[b] No lump-sum return of emission charge

Figure 1: Key elements of the integrated farm-level model

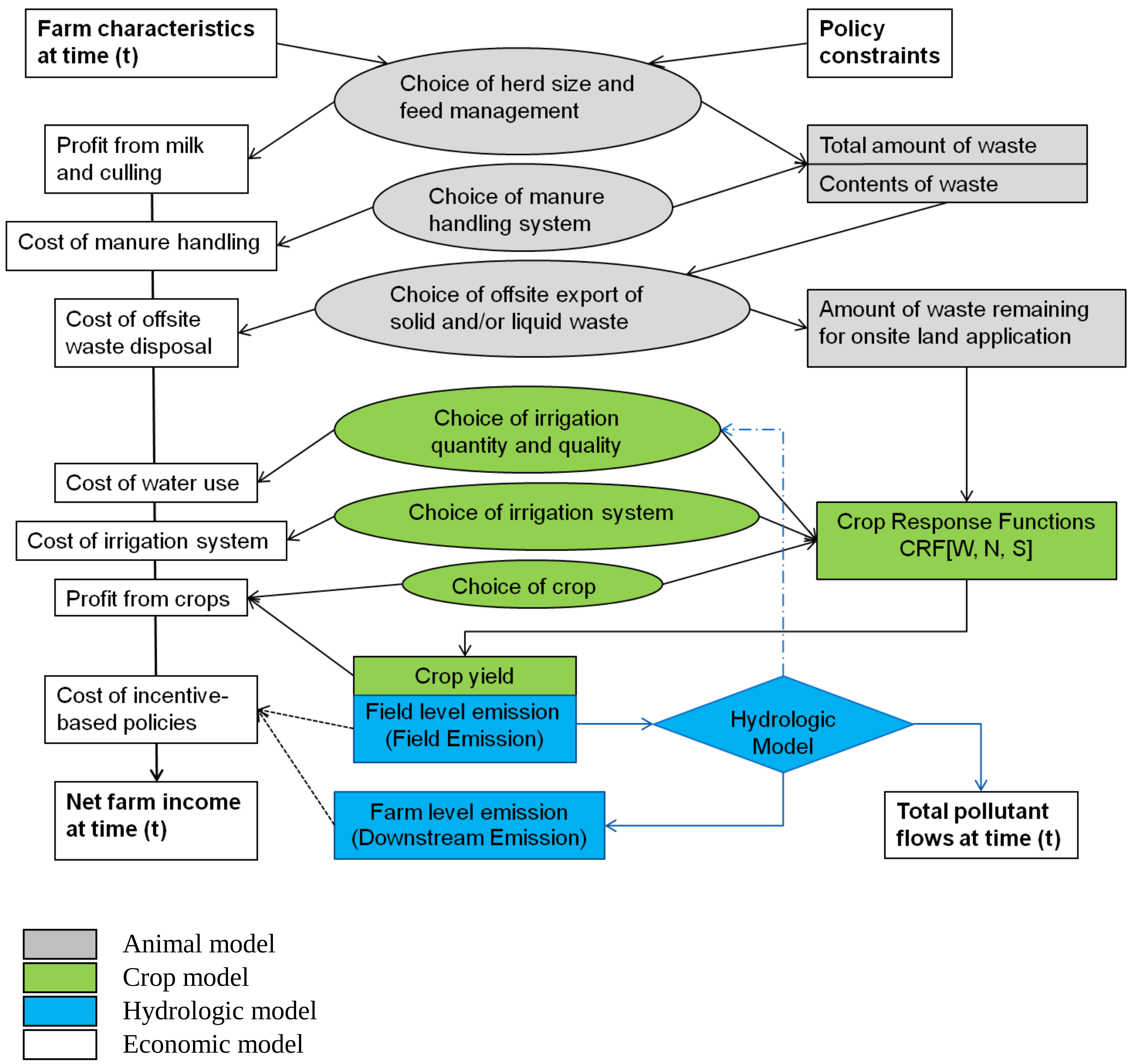

Figure 2: Baseline scenario (flush-lagoon, 1/4-mile furrow, corn-wheat rotation). (a)-(c): paths of soil organic nitrogen, soil inorganic nitrogen, and soil salinity for each field type. (d): paths of irrigation for each water source. (e): paths of field emission for each field type. (f): paths of total field emission and downstream emission.

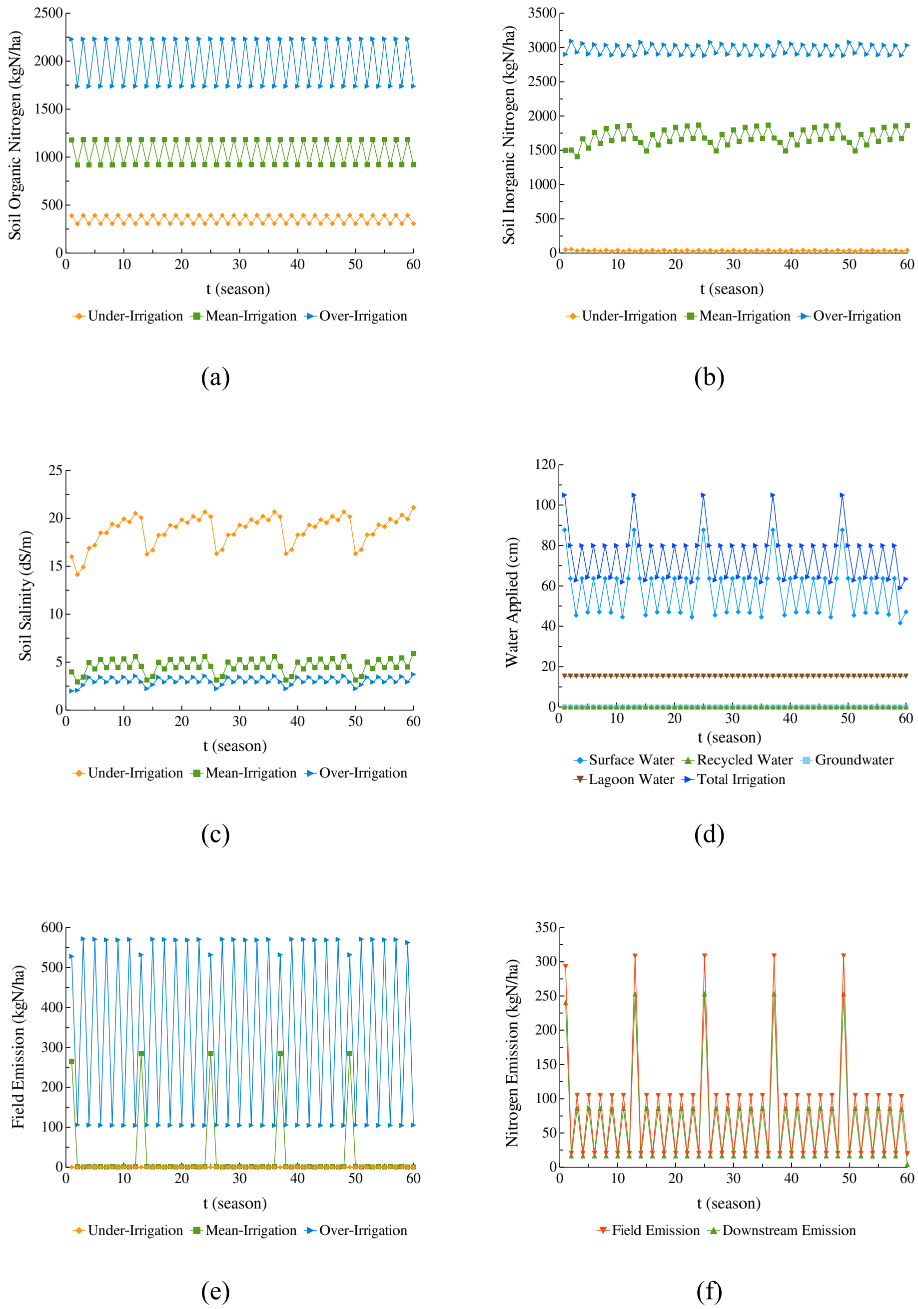

Figure 3: Nutrient Management Plans scenario with the optimal activity set (scrape-tank, 1/4-mile furrow, corn-wheat rotation). (a)-(c): paths of soil organic nitrogen, soil inorganic nitrogen, and soil salinity for each field type. (d): paths of irrigation for each water source. (e): paths of field emission for each field type. (f): paths of total field emission and downstream emission.

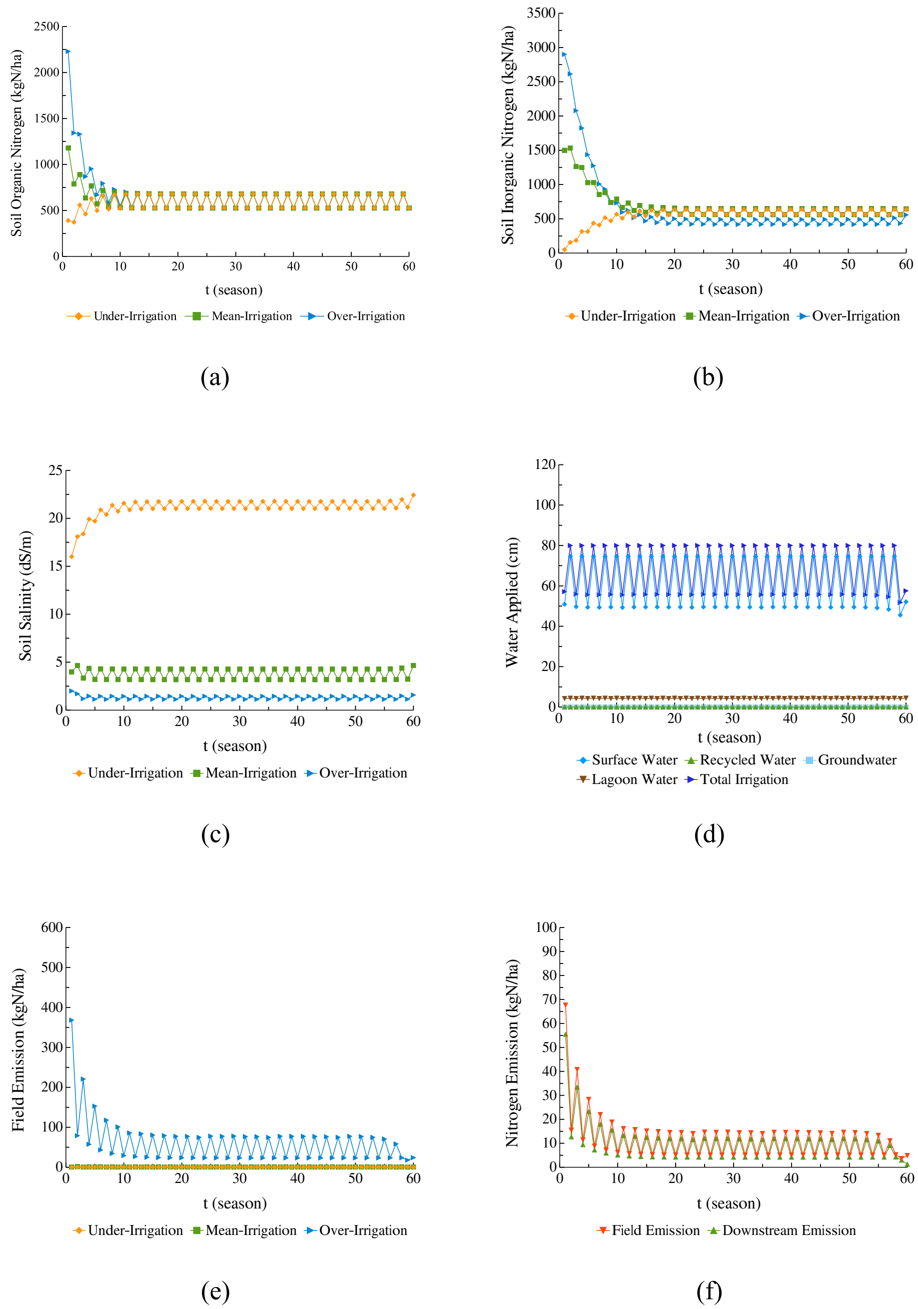

Figure 4: Field emission limit scenario with the optimal activity set (flush-lagoon, linear move, and corn-wheat rotation). (a)-(c): paths of soil organic nitrogen, soil inorganic nitrogen, and soil salinity for each field type. (d): paths of irrigation for each water source. (e): paths of field emission for each field type. (f): paths of total field emission and downstream emission.

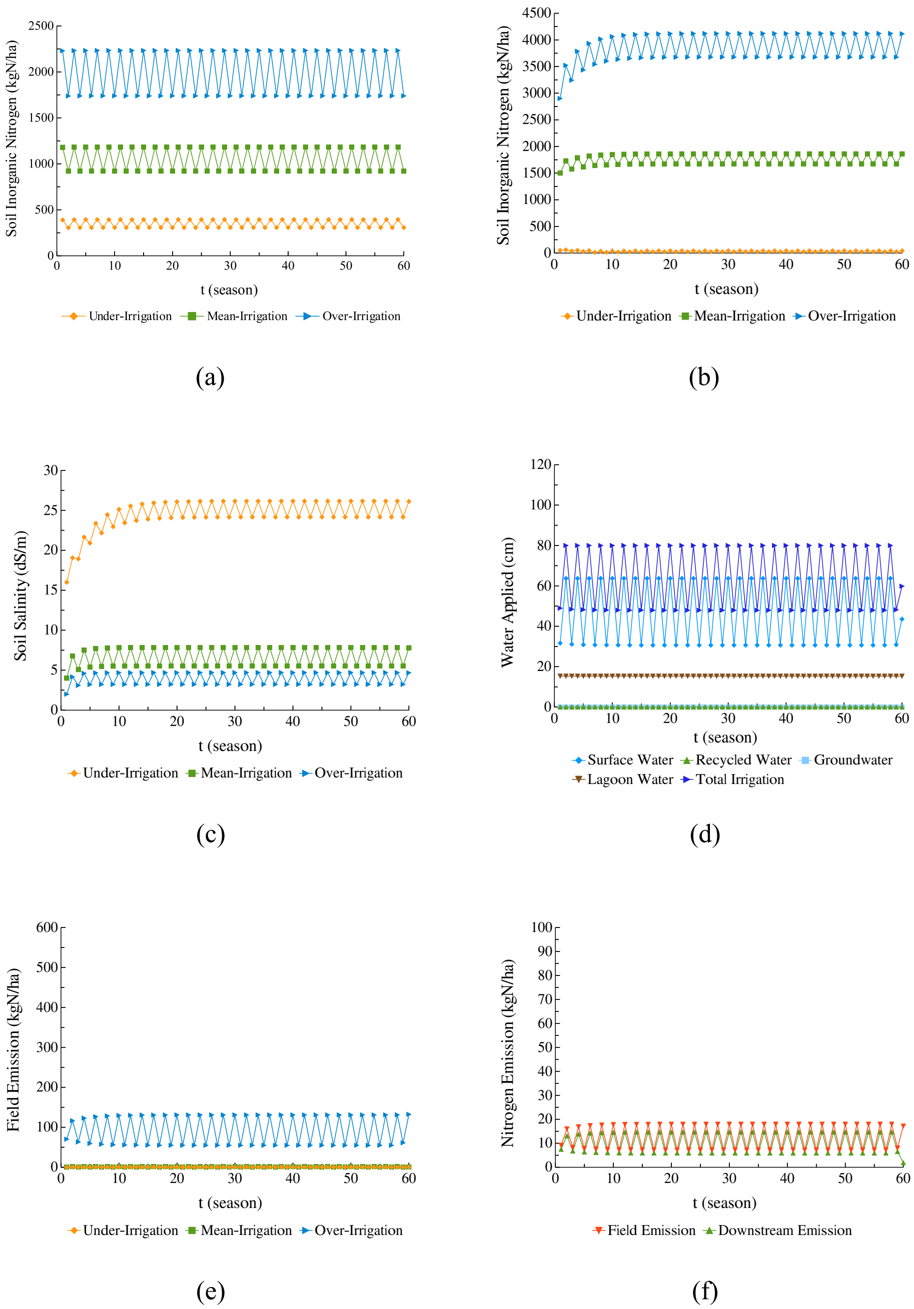

Figure 5: Downstream emission limit scenario with the optimal activity set (flush-lagoon, 1/4-mile furrow, corn-wheat rotation). (a)-(c): paths of soil organic nitrogen, soil inorganic nitrogen, and soil salinity for each field type. (d): paths of irrigation for each water source. (e): paths of field emission for each field type. (f): paths of total field emission and downstream emission.