

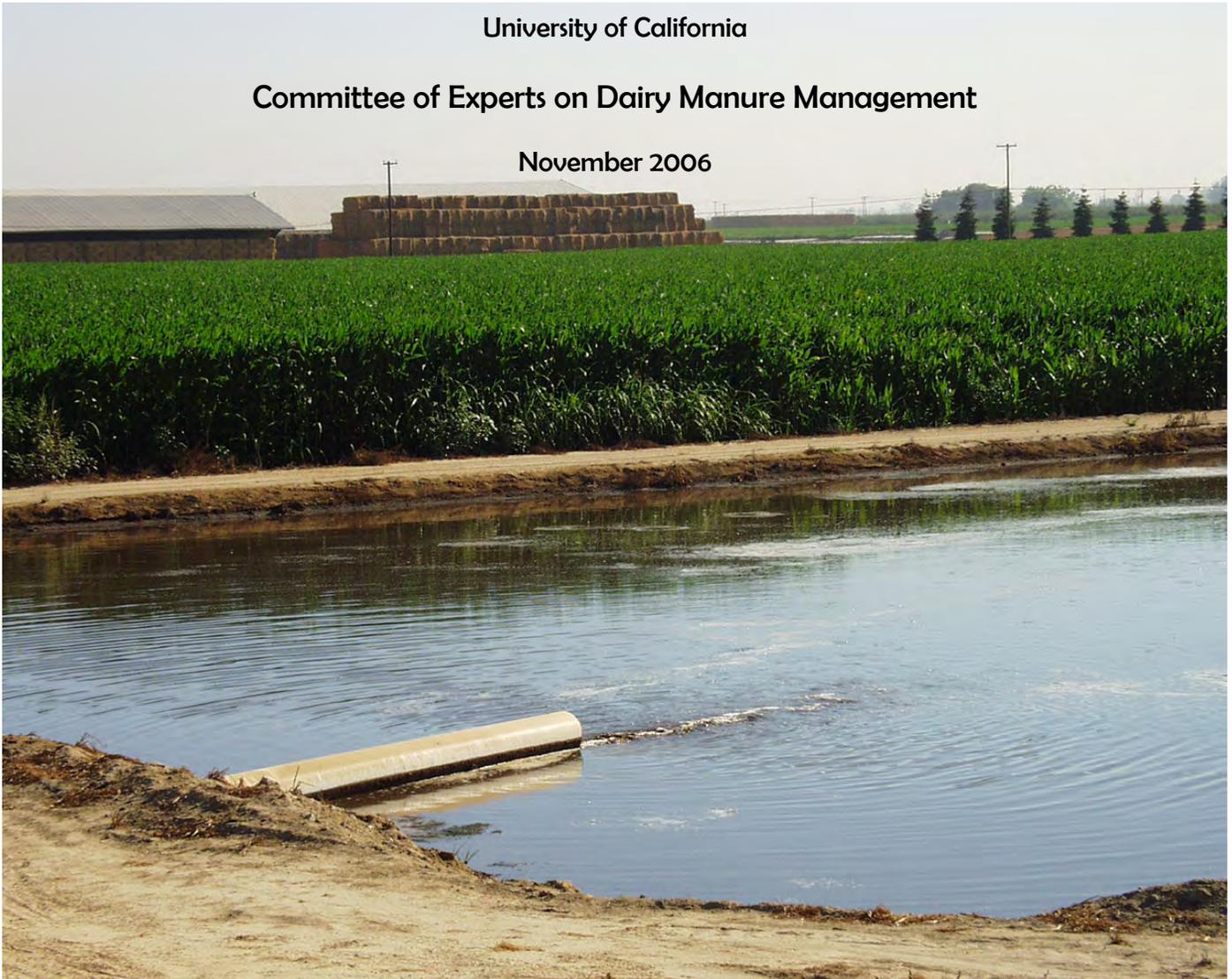
Groundwater Quality Protection:

Managing Dairy Manure in the Central Valley of California

University of California

Committee of Experts on Dairy Manure Management

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Executive Summary

1. Introduction

There are approximately 1.7 million lactating dairy cows in California. Sale of milk from these cows is estimated at \$4.6 billion of the state's \$30 billion agricultural market. Nearly 73% of the cows are located in the San Joaquin Valley, which consists of San Joaquin, Stanislaus, Merced, Madera, Fresno, Kings, Tulare and Kern counties. The Central Valley Regional Water Quality Control Board submitted a list of questions to Vice President Gomes requesting specific information related to dairy manure. The answers to these questions are the basis for this report.

The responsibility of the Committee was to answer these questions and, in areas where data were not available, conduct research (if short term research would provide insight) or work toward long term research objectives to provide more complete answers for the future. The following lists the original six groups of questions with a summary answer of the committee to these questions.

2. Nitrogen Excretion

- A. How much N is excreted by the average lactating dairy cow?
- B. How much N is excreted by dry stock?
- C. Are there regional differences in California and, if so, what factors should be applied to design criteria for lactating and dry cows due to the geographic area of the dairy?
- D. Does diet play a role in the amount of N excreted and, if so, what are the adjustments that should be made in N generation rates based on various common diets?

The recommended average excretion of N by lactating dairy cows in California is 462 g N/head/day, and by dry cows it is 195 g/head/day. However the Committee stresses that these are average numbers, that the trend will be to higher values in the future as milk production per cow increases, that actual values will vary sharply among dairies, and that there are techniques available to make very accurate site specific estimates.

Short of measuring feed intake N, which allows for highly accurate estimates of manure N excretion using, e.g., the UC Dairy Animal Waste Model, the Committee believes that expressing manure N excretion relative to milk production, rather than body weight or 'animal units', is the biologically most sensible assessor on within site efficiency of animal N use, as manure nutrient production is primarily a function of feed intake and feed intake is primarily a function of milk yield. There is no evidence that regional differences in California are of a sufficient magnitude to require additional region specific design criteria for dairy farms.

Dairies feeding better formulated diets will tend to have higher efficiencies (i.e., the lowest N excretion relative to milk production) while poorly managed dairies, where cows may be fed poorly formulated diets, will tend to have lower efficiencies (i.e., the highest N excretion relative to milk production). In contrast, the better managed dairies with higher per cow milk production

will tend to have the highest manure N excretion per cow, thereby demonstrating the error in expressing manure N excretion on a per cow basis, rather than on the basis of milk produced.

There is no evidence that diet formulation principles that are based upon nutrients differ among regions of California, even if the cost effective access to different feedstuffs varies among regions. Hence, different 'region typical' diets need not be considered.

3. Manure Distribution

Three categories describe most cow and manure management systems in California, these being freestalls with flush systems, corrals with flush alleys, and corrals with no flush alleys. Dairies frequently have more than one type of management system.

- A. What percentage of each type of waste enters the retention ponds, and what part remains in corrals, under the three operations?
- B. How much N is carried with rainwater from the corrals into the liquid waste storage?

Manure distribution patterns vary depending on the facility infrastructure and operational and managerial decisions. Most likely, the manure distribution pattern will not be the same in any two dairies. Even within the same dairy, the deposition pattern shifts as operational parameters change. The amount of manure collected in liquid will vary from 8% (only manure excreted in the milking parlor) to 100% (manure from animals always living on concrete). More intensively managed systems (freestalls) will collect 42 to 100% of manure on a daily, monthly, or yearly basis.

There are no specific data to quantify the nutrients carried from the corral to the retention pond as a result of rain runoff. Data from Florida suggest this number is minimal.

Solids removal from mechanical and gravity flow separation systems are quite poor (under 25%) to good (consistently 50%). However, nutrient removal cannot be described based on a percent of solids removed. Soluble nutrients and salts predominantly remain in the liquid system.

4. Nitrogen losses

- A. How much N is lost between generation and land application of the liquid waste under different waste handling scenarios?
- B. How much N is removed with solids separation, and is there a difference when mechanical as opposed to gravity separators are used?
- C. How do factors such as frequency of flushing, recycling of water for flushing and retention time in ponds affect losses?

Use of a universal animal 'emission factor' for reactive N compounds (e.g., ammonia) from commercial dairies is not possible because of the limited number of field measurements on which they are based and the wide emission variability among and within dairies. We concur

with the National Research Council (NRC, 2003), that there is no single emission factor that could possibly describe atmospheric N losses from dairies.

To determine atmospheric N losses from existing dairies, we concur with NRC (2003) in recommending the use of process-based models coupled with a whole-farm and farm component N balance approach that describes potential atmospheric N losses for the different farm component processes. The process-based model approach coupled with total N balance predicts atmospheric N losses between excretion and land application and helps manage nutrient application rates to crops at agronomic rates. The Committee emphasizes however, that while this approach is technically viable, it requires extensive data measurement, record keeping, and is associated with significant estimation errors.

There are insufficient data available to quantify atmospheric N losses associated with effects of frequency of flushing, use of recycled water for flushing and time spent in retention ponds. However, in general, more frequent flushing, of fresh water for flushing and shorter residence times in lagoons will tend to decrease ammonia volatilization. However the quantitative impact of these strategies is unknown at this time.

In light of these findings and in light of California-specific conditions, we suggest that atmospheric N losses from liquid manure (i.e., freestalls and flush lanes and lagoons) used for dairy planning and permitting purposes, are considered to range between 20% and 40%. The use of a single number (“emission factor”) is strongly discouraged. Note, that these losses do not include atmospheric N losses in the land application (crop production) area.

5. Crop Nitrogen Requirements

- A. Is the Western Fertilizer Handbook appropriate to determine crop nitrogen requirements on California dairy farms?

Both field and modeling studies reviewed and implemented for this report consistently show that the N input requirements for forage crops will generally be in the range of 140% to 165% of the crop N harvest removal, assuming that the manure application would consist of lagoon water which is approximately 75% $\text{NH}_4\text{-N}$. As stated above, inputs include not only manure and fertilizer N but also atmospheric N sources and nitrate present in irrigation water. Investigations of the crop N recovery in several field experiments showed that the appropriate N loading rate that minimizes N leaching and maximizes N harvest is between 140 to 150% of the N harvested. Computer models indicated a somewhat larger range of 140% to 165%. While field studies provided important feedback on loss pathways and loss rates as well as mineralization rates, model simulations were well suited to study the dynamic behavior of the soil nitrogen pool and its interaction with the crop N uptake. Simulations are particularly valuable to understand the role of various loss pathways. Field mineralization, volatilization, and denitrification rates for specific field conditions can be obtained from detailed field and laboratory studies using standard model calibration and validation approaches.

The combined evidence from laboratory, field, and modeling studies indicates that precise nutrient management, while plausible in principle, may be problematic when implemented in full-scale production systems, as it requires careful timing of the N applications, close monitoring of the amount of N and water inputs, and best management of crop production. More importantly, the growers must show flexibility to make necessary adjustments on N inputs during the course of a growing season to achieve satisfactory results.

With respect to the potential for groundwater degradation, all of the computations and field observations point to a fundamentally critical issue: Given that practically achievable leaching fractions in border check and furrow systems are 15% to 30%, nitrate leaching is at best in the range of 10% to 15% of the N applied. The corresponding nitrogen input requirement is 140 to 165% of N removal at harvest. At annual crop yields that typically remove 400 – 600 lbs N ac⁻¹ yr⁻¹, input requirements will be in the range of 560-990 lbs N ac⁻¹ yr⁻¹. Hence, nitrate-nitrogen leaching losses – under optimal irrigation and nutrient management – will be in the range of 55 to 150 lbs N ac⁻¹ yr⁻¹. Assuming recharge rates in irrigated systems of 1 – 2 acre-feet per acre per year (300 – 600 mm per year), the nitrate concentration in the leachate is in the range of 10 to 55 ppm (mg L⁻¹) NO₃-N, which is at or above the regulatory limit for drinking water quality (10 mg L⁻¹) and at or significantly above the average measured leachate value for other California farming systems (15 mg L⁻¹, Rible et al. 1979). The potential for denitrification in the unsaturated zone below the root zone (not considered in this report) and within the Central Valley aquifers therefore becomes a key factor in determining, whether such (optimal) leaching water quality conditions will still cause groundwater degradation or whether denitrification naturally attenuates nitrate levels to non-degrading levels.

6. Phosphorus and Potassium Requirements

- A. Should the application of phosphorus and potassium be limited?
- B. If so, what should the limits be and under what circumstances?

Currently available literature data and the limited amount of recent research in the San Joaquin Valley on the nutrient cycle of P and K in dairies does not allow for a conclusive finding on the question of phosphorus and potassium status in field soils used for application of animal waste. However, trends are certainly apparent that dairymen will need to monitor the increased application of both P and K to determine the impact on forage nutrient levels and animal performance.

7. Salt Excretion and Land Application of Salt

- A. How much salt is generated by a cow?
- B. Are there differences in the amount of salt generated between dry and wet stock?
- C. How much salts are removed by various solids separation methods?
- D. What are “reasonable” salt loading rates in typical Central Valley cropping patterns?

The excretion of salts can be expected to vary dramatically due to on-farm management decisions and practices. Regardless, daily consumption and excretion of salts will be dramatically lower for dry vs. lactating stock, although even this can be affected by management decisions.

From a dietary point of view, exact salt excretion data are currently unavailable except for Na^+ , K^+ , Cl^- , and total N excretion. Excluding uncontrolled provision of salt to lactating dairy cows, a summary of six groups of cows on three commercial California dairies suggests that the average lactating dairy cow will excrete 1.29 lb (585 g) day^{-1} of Na^+ - K^+ - Cl^- salts and the average dry cow will excrete 0.63 lb (287 g) day^{-1} (also Na^+ - K^+ - Cl^- salts only). Assuming an annual division of 305 days lactating and 60 days dry, the average dairy cow will excrete 427 lb (194 kg) year^{-1} of Na^+ - K^+ - Cl^- salts. That these values are less than values reported by the 1973 UC Committee of Consultants Water Quality Task Force reflects the fact that comparable data for other salts (Ca^{2+} , Mg^{2+} , HCO_3^- , SO_4^{2-}) are not available.

Analysis of manure data, geochemical modeling, and observations of groundwater recharge quality in the San Joaquin Valley dairies suggests that the salinity contribution (defined as the mass of total dissolved solids) from manure, under proper nutrient management practices that seek to maximize the use of lagoon water as a source of fertilizer, is on the order of 1786 – 3572 $\text{lbs ac}^{-1} \text{ yr}^{-1}$ (2000 – 4000 $\text{kg ha}^{-1} \text{ yr}^{-1}$). For comparison, the salt loading from irrigation water alone, depending on the source of the irrigation water, is on the order of 357 $\text{lbs ac}^{-1} \text{ yr}^{-1}$ (400 $\text{kg ha}^{-1} \text{ yr}^{-1}$) for lower salinity water sources (e.g., Sierra Nevada watersheds) to nearly 4,000 $\text{kg ha}^{-1} \text{ yr}^{-1}$ for higher salinity water sources (e.g., State Water Project).

At the regional scale, dairies are only one of several sources of salinity to the Central Valley's groundwater and surface water supply. Locally, they add significant additional salinity to groundwater. The long-term impacts from dairies as well as those from other salinity sources (municipal wastewater treatment plants, food-waste dischargers, etc.) are still not well understood. Increasing salinity in California's waters is an issue that must be dealt with as part of an integrated long-range water resources management plan.

Chapter 1 - Introduction

1.1 The Dairy Industry in California

Dairying is an important agricultural industry in California as it is the number one dairy state in the nation with approximately 1.7 million lactating cows. As of December 2003, the state's output in milk and cream accounted for approximately \$4.6 billion of California's nearly \$30 billion in annual agricultural income.

1.2 Regulatory Framework

Federal, State and Local regulatory agencies can regulate dairy facilities. The Central Valley Regional Water Quality Control Board and the San Joaquin Unified Air Pollution Control District are the primary agencies. County agencies in San Joaquin, Merced, Madera, Kings, Tulare and Kern also regulate dairy facilities.

Multiple efforts are underway to address environmental concerns related to animal confinement at the federal, state, and local level. The US EPA promulgated the Concentrated Animal Feeding Operations (CAFO) Final Rule on Dec. 15, 2002, and it was published in the Federal Register on Feb. 12, 2003. US EPA identified existing and potential concerns during the Rule making process. The new Rule required that all CAFOs apply for an NPDES permit by April 13, 2006. The Rule was challenged in court and the Judges' decision vacated the duty to apply. Only dairies with surface water discharges are obligated to seek coverage with an NPDES CAFO specific permit. Permit details are anticipated to be available in spring 2007 with a duty to apply deadline of July 31, 2007. The Central Valley Regional Water Quality Control Board is developing guidelines under both federal (NPDES permit) and state regulations. California regulations governing discharges from confined animal facilities are contained in Title 27 of the California Code of Regulations, which was established by the Porter-Cologne Water Quality Control Act (California Water Code Division 7).

The San Joaquin Valley Unified Air Pollution Control District adopted its implementation plan for SB 700 on May 20, 2004, and some dairy operators were required to request coverage under the new permit structure. Most dairy operators were required to develop and implement Conservation Management Practices to reduce fugitive dust emissions (Rule 4550). The South Coast Air Management Control District has finalized Rule 1127 and will begin the implementation phase to reduce dust emissions.

Key to many of the regulations is the need to develop manure management plans to include environmental compliance. Though not a regulatory agency, the USDA Natural Resources Conservation Service is developing Guidance for California's Comprehensive Nutrient Management Plan (CNMP). This process would provide key information for consultants and producers to adequately document nutrient application and crop nutrient uptake and identify management practices to reduce atmospheric emissions.

1.3 Environmental Concerns of Regulatory Agencies

Manure generated in dairy production facilities can impact the environment (National Center for Manure and Animal Waste Management, 2001). Water quality impacts from nitrate and salts from land applied manure are key concerns to the Central Valley Regional Water Quality Control Board and are the catalyst for this report. Emissions from manure deposited in corrals, holding pens, and lagoons may also have adverse effects on air quality. Local air districts are concerned with ammonia, volatile organic compounds (VOCs), hydrogen sulfide and particulate matter (PM) emissions. Methane emissions occur but are not a regulated compound.

Potential water quality impacts from improper handling and disposal of dairy manure include accidental or intentional discharges or inadequate management of nutrients that cause pollutants to reach surface water or to recharge to groundwater. Other concerns related to the potential contribution of manure discharge to surface waters were included in the CAFO rule, and include microbes, antibiotic metabolites and organic compounds (US EPA CAFO Rule, 2003).

The San Joaquin Valley Air Basin (SJVAB) is not in compliance with the Clean Air Act. For ozone, the US EPA reported the basin's air quality to be in the "Non-attainment/Extreme" category, and for PM₁₀, in the "Non-attainment/Serious" category. In general, ozone is not emitted to the air from specific sources, but its precursors VOCs and NO_x are emitted. Ozone is the product of photochemical reactions of N oxides (NO_x), and VOCs. These pollutants are emitted from dairies at levels that have not been studied conclusively. Ruminant animals generate methane and certain VOCs (e.g., volatile fatty acids, ketones, aldehydes, alcohols) as a part of their normal digestive processes. Additionally, methane is generated under anaerobic conditions in liquid manure storage ponds. There are other organic gases produced from manure and its enzymatic or microbial decomposition. Ammonia released from dairy manure is odorous and can react with NO_x to form ammonium nitrate, which is classified as fine PM. Ammonia is also important because it can react with oxygen in the atmosphere and the soil to form NO_x and eventually nitrate.

1.4 The Role of the University of California

In 1973, the University of California, Division of Agriculture and Natural Resources, at the request of State Water Resources Control Board, formed the Water Quality Task Force of UC Agricultural Committee of Consultants (1973) to investigate generation of manure on dairies and make recommendations on management of dairy manure in California. The documents produced formed the basis for development of statewide regulations adopted in 1984.

The Central Valley Regional Water Quality Control Board (CVRWQCB) asked the University of California to establish a committee to review the relevant issues. Dr. Reg Gomes, Vice President of the UC Division of Agriculture and Natural Resources convened a Dairy Manure Management Committee of University experts to address technical questions posed by the staff of CVRWQCB. Committee members were:

- Andrew Chang, Associate Director, Center for Water Resources, University of California

- Thomas Harter, Associate CE Groundwater Hydrology Specialist, Department of Land, Air, and Water Sciences, University of California, Davis
- John Letey, Director, Center for Water Resources, University of California
- Deanne Meyer, CE Livestock Waste Management Specialist, Department of Animal Science, University of California, Davis
- Roland D. Meyer, CE Soil Fertility Specialist, Department of Land, Air, and Water Sciences, University of California, Davis
- Marsha Campbell-Mathews, Farm Advisor, UC Cooperative Extension Stanislaus County
- Frank Mitloehner, Assistant CE Air Quality Specialist, Department of Animal Science, University of California, Davis
- Stu Pettygrove, CE Soils Specialist, Department of Land, Air, and Water Resources, University of California, Davis
- Peter Robinson, CE Dairy Nutritionist, Department of Animal Science, University of California, Davis
- Ruihong Zhang, Associate Professor, Department of Agricultural and Biological Engineering, University of California, Davis

The committee met from September 2001 through June 2005 to deliberate these issues:

- Amounts of manure and N excreted by dairy cows
- Distribution of excreted manure and N in dairies
- Loss of N during storage
- Fate and transport of manure N and salts following application on cropland

Committee members conducted field monitoring and experiments to collect relevant data. Models were developed to simulate crop yields and nitrate leaching when dairy manure is applied on cropland for forage production.

In the course of the investigation, the Committee was assisted by many technical support staff of the University of California, including:

- David Birkle, Staff Research Associate, Center for Water Resources, University of California
- Gounglon Feng, Post Doctoral Research Scientist, Department of Environmental Sciences, University of California, Riverside

We also acknowledge Chris Amrhein, Ken Tanji, and Alejandro Castillo for their critical comments on portions of this report.

1.5 The Role of this Report

This report summarizes the current scientific literature and research results and provides – to the degree possible - answers to the original questions posed.

In addressing the questions, the committee needed to acknowledge that dairies and their environment are inherently complex systems. At a minimum, we distinguish the following systems within the dairy-environment complex:

- animal system
- animal feed system
- animal housing system
- animal health system
- milk production and food safety system
- manure collection, treatment, recycling, and storage system
- manure application system
- forage crop production system
- irrigation system
- the atmospheric environment system
- soil environment system
- surface water and groundwater environment systems

These systems are tightly linked and closely interact with each other. Yet, each has classically been described, investigated and researched by specific disciplinary researchers in animal science, agronomy and engineering. Only recently have multi-disciplinary approaches been implemented to better understand and characterize the interactions of dairy systems with their land, air and water environments. Addressing the CVRWQCB questions took efforts of a multi-disciplinary team, and most of the research that has been reviewed has been implemented in a disciplinary (i.e., single system) fashion. However, we attempted to interpret and summarize literature relevant to the questions that were posed. In the process, we found that our scientific and technical language, as well as the multitude of methods used among the various disciplines involved, were not always compatible thereby forcing careful interpretation of available information. Needless to say, such interpretation is not unique. In this document, we have attempted to leave behind disciplinary bias as much as possible and provide the best possible answers given the background and experience of the committee members. Clearly, the answers may not be the only possible answers – depending on the point of view; and some can be formulated in either a much narrower or in a more general way. Hence, we expect both, constructive critique and disagreement with the results provided.

Most importantly, we find that these questions touched on an interdisciplinary field of research around animal farming and environmental issues that is only now being established. New results are being published almost daily. This document reflects our knowledge as of early 2005. We expect that some of the questions will need to be re-examined within less than 5 years, while the answer to others may not change for a decade or more. We fully understand that this poses a critical challenge to the regulatory agencies and to the dairy industry, who urgently seek long-term solutions (rather than intermediate fixes) to addressing environmental issues on dairies. Finally, the peer-reviewed UC ANR publication (2006 revision) is identical to the 2005 report submitted to the CVRWQCB, except for minor editorial changes to clarify text, improve spelling, grammar, and consistency throughout the report. We appreciate the extensive comments from four anonymous technical reviewers and the work of UC ANR associate editor, Dr. Jay Gan.

Chapter 2 - Manure and Nitrogen Excretion

2.1 Introduction

Amounts of manure and nitrogen (N) excreted by dairy cows have been reported in the literature since the 1960s, although the values have not been consistent. These discrepancies create uncertainties in establishing and implementing dairy manure management plans. In a 1973 correspondence, the Water Quality Task Force of the UC Committee of Consultants concluded that an average dairy cow in California produced manure N at the rate of 0.4 lb/head/day (182 g N/head/day). This conclusion was reached by conducting feed-milk-manure N mass balance calculations, and by chemical analyses of feed and milk samples collected in the state, thereby accounting for the N content of feeds that are in dairy rations and the N content of milk produced. The N excreted by a dairy cow was the net difference of N input (i.e., amount of N in feed consumed) and N output (amount of N in milk produced), and mass balance estimates were used to derive estimated total manure and N excretion for dairy cows in the Santa Ana Basin, Central Valley and North Coast regions (Table 2-1).

Table 2-1: 1973 Water Quality Task Force of the UC Committee of Consultants Estimate of Total Manure and N Excretion of Dairy Cows in California. N excretion was obtained from the difference between intake and output. Although physiologically inaccurate (some of the N goes to animal growth), it is a reasonable approximation for adult animals.

Region	Ration	Dry Matter (lb/head/day) ⁺			Nitrogen (lb/head/day) ⁺			N (% of intake)
		Intake	Output in Milk	Intake minus Output	Intake	Output in Milk	Excretion	Excretion
Santa Ana	Alfalfa Forage	42.84	6.77	36.07	1.14	0.29	0.85	74
Central Valley	Multiple Forage	42.42	6.77	35.65	0.90	0.29	0.61	67
North Coast	Cereal Forage	44.00	5.86	38.14	0.98	0.25	0.73	74

⁺1 lb/head/day equals 0.454 kg/head/day¹ in SI unit. "Head" refers to a 1,400 lbs adult Holstein cow.

The 1973 Water Quality Task Force of the UC Committee of Consultants found that the N excretion values estimated by their mass balance method "exceed the actual amount excreted by about 0.4 lb/cow/day as determined by chemical analyses of the wastes" and reconciled the discrepancy between the two estimates by concluding that "the difference is believed to be due to denitrification, within the ruminant animal, of a substantial portion of the input N".

Thus the N excretion value recommended by the 1973 Water Quality Task Force of the UC Committee of Consultants was approximately 50% of the mass balance calculated N excretion. The assumption that much of the ingested N was lost through ammonia volatilization from, or

denitrification within, the intestinal tract of the cattle could not be confirmed based upon the technical literature. Indeed, the rumen of dairy cows is not a conducive chemical environment for oxidation of organic N compounds in feed to nitrate, the precursor of denitrification, and unless the N was already in an oxidized form, precursors for denitrification (i.e., nitrate and nitrite) would have been absent.

Since 1973, the nutritional management of dairy cows has improved in California. Advances in dairy nutrition, genetics and management have resulted in slightly larger cows and sharply increased milk production. With the changes in rations, the quantity and chemical composition of manure were expected to change. Tomlinson et al. (1996) demonstrated that N excretion by dairy cows could be predicted from dietary intakes of dry matter and its N content. Based on a mass balance of dietary N intake and the N content of the milk, Van Horn et al. (1998) estimated that N excretion by lactating cows varied from 0.73 to 0.85 lb/head/day (330 to 384 g/head/day), figures that are in general agreement with values frequently reported in the literature (e.g., Arago et al., 2001).

In this chapter, we review currently accepted methods for estimating nutrient excretion from dairy cows (section 2.2), apply these methods to develop best estimates of average excretion rates at California dairy farms (section 2.3), discuss regional differences in excretion rates (section 2.4), and summarize the role of diet on excretion rates (2.5).

2.2 Methods to Estimate Manure Nutrient Excretion by Cows

At least three methods can be used to estimate manure and nutrient excretion by groups of dairy cows, although the choice of the most appropriate method depends on how the results will be used. For example, calculations made from newly revised Standard Table D384.2 (American Society of Agricultural and Biological Engineers - ASABE) are most useful when planning a new dairy (i.e., when no cows actually exist). Whole farm mass balance models, on the other hand, document nutrients entering and exiting the entire dairy operation (i.e., document the whole farm balance but provide no information on nutrient balances in any sector of the facility). Finally, estimates based on farm sector mass balance models¹ that use production, diet and physiological status of defined groups of cattle within the dairy are useful in documenting nutrients entering and exiting the entire dairy operation while allowing evaluation of the nutrient balances for any defined sub-group of cows (i.e., are useful in identifying sectors of the dairy where efficiencies may be low and so allow dairy operators a means to improve overall dairy operation efficiency). The following discussion provides brief reviews of these approaches.

2.2.1 Method 1. Calculations made with Standard Table D384.2 of ASABE

Historically, manure excretion has been reported relative to body weight. However for lactating cows, it is far more appropriate to report excretion on per unit of milk produced as milk production of cows varies to a much greater extent than does body weight, and because cows eat

¹ Note that, in the context of estimating N excretion, the terms “loss” and “mass balance” refer to the animal system itself, not to other environmental systems such as atmospheric or groundwater losses from lagoons or N fluxes from the production area to the land application area. These are discussed in later chapters of this report.

more feed as their milk production increases. In addition, breed differences are much smaller when manure and nutrient excretion are expressed per unit milk produced.

ASABE has revised its standards for manure production based on data collected from total manure collection studies with cows at numerous US research facilities. Regression analyses were conducted for manure and N excretion for lactating cows, dry cows, replacement heifers and young calves, in order to create tabular values.

For example, with the new ASABE equations, estimates of dry manure and N excretion for an ‘average’ California lactating dairy cow weighing 1375 lbs and producing 88 lbs of milk/day while consuming 1.37 lbs of N/head/day, were 19.5 lbs and 0.97 lbs/head/day, respectively. Similarly, using the new ASABE equations for a California ‘average’ dry dairy cow weighing 1660 lb and consuming 0.56 lb of N/head/day, dry manure and N excretion was estimated to be 10.8 lbs and 0.49 lb of N/head/day, respectively.

The ASABE approach is extremely useful as a means to estimate future N balances when planning a new dairy (i.e., when no cows actually exist). Once the dairy has been built, actual production levels may vary and result in substantially higher or lower excretion levels than projected from the ASABE method (see, for example, Table 2-2).

2.2.2 Method 2. Estimates based on whole farm mass balance models

Whole farm mass balance models (with respect to the animals) are most useful in documenting manure nutrient outputs from entire commercial dairies. These mass balances are done by documenting total N inputs and exports from the animals in the animal operation (Koelsch, 2001). The approach requires the dairy facility operator to measure and quantify all sources of N entering and leaving the animal operation. Data can be obtained from records of feed purchases as well as sale of cattle and milk.

From a regulatory perspective, such animal-based mass balance models (e.g., Meyer and Robinson, 2002) offer a site specific method to quantify the N excretion in a dairy operation, but provide no information on the nutrient balances in any animal sector of the facility since the evaluation is based upon a complete input/output balance of all nutrients to/from an entire dairy herd.

The whole farm mass balance approach offers little insight as to why a particular facility may be out of compliance, as it provides no information on the efficiency of the definable and measureable components of the facility, and none at all on how to regain compliance.

2.2.3 Method 3. Estimates based on farm sector mass balance models

Farm sector mass balance models (individual mass balances on each identifiable group of cattle on the dairy) are useful in documenting manure nutrient, including N, outputs from entire commercial dairies as the sum of its identifiable cattle groups. This approach requires the dairy facility operator to measure and quantify feed characteristics, physiological conditions of the cattle, and the conditions of the cattle’s environment for all identifiable groups of cattle on the

dairy. Data can be obtained from herd records, observations of cows, climatic information databases and feed analyses. Animal mass balance outputs, by cattle group, are based on consumption of dry matter, energy, N and minerals. Estimates of the nutrient content of feces and urine are calculated by subtracting nutrients partitioned into product, growth and reproduction from nutrients consumed.

From a dairy management perspective, such models (e.g., SHIELD; Robinson, 2000) offer a site specific way to quantify the N balance and N excretion on a dairy operation, while providing information on (animal) nutrient balances in each cattle sector of the facility, since the evaluation is based upon complete input/output balances of all nutrients to/from each defined cattle sector of the dairy facility. An example of such site-specific calculations is provided in Appendix A.

The farm sector mass balance approach will likely be the most useful way to provide information to a dairy operator on the nutrient efficiency of the various cattle sectors of the dairy, thereby identifying the sectors with the poorest efficiency. These can be targeted for improvement by the dairy.

2.3 UC Committee of Consultants Estimate of Nitrogen Excretion by Dairy Cows

The UC Dairy Animal Waste Model was developed from SHIELD to estimate whole dairy manure nutrients as the sum of those of its cattle groups. Additional nutrients (e.g., P, K, Ca) and trace elements (e.g., Zn, Cu, and Se) in the manure are also estimated. To create estimates, rations used were the measured dietary intakes of high and low production strings of lactating dairy cows in four representative commercial dairies in four regions of California, being Petaluma, Modesto, Tulare and Chino. The measured feed compositions, cow characteristics and milk productions were incorporated into the evaluation.

Based upon on-farm measurements and evaluation using the UC Dairy Animal Waste Model, these lactating cows excreted 0.92 to 1.25 lb N/head/day (417 to 565 g N/head/day), depending on the rations of the cows on the specific dairies (Table 2-2). The average N excretion was 1.02 +/- 0.13 lb/head/day (462 +/- 60 g N/head/day).

These estimates are considerably higher than the 1973 Water Quality Task Force of the UC Committee of Consultants estimate of 0.61 to 0.85 lb N/head/day (average 0.69 +/- 0.18 lb N/head/day), or 277 to 386 g N/head/day (average 315 +/- 81 g N/head/day). The increases primarily reflect increases in milk production, and the resulting increase in feed intake, between 1973 and 2004.

In apparent contrast to these estimates, a white paper prepared for the National Center for Manure and Animal Waste Management (Arogo et al. 2001) estimated that the daily average N excretion of dairy cows in the US varies from 0.58 to 0.82 lb N/head/day for lactating cows weighing 992 to 1,400 lb (263 to 372 g N/head/day for lactating cows weighing 450 to 635 kg). However milk production levels of dairy farms in California are higher than the national average,

and so N excretion would also be expected to be higher than the national figures of Arogo et al. (2001).

Table 2-2: N Balance of Lactating Dairy Cows in California.

Region/ Ration ⁺	Intake (lb/day) ⁺	Milk (lb/day) ⁺⁺	Body Change (lb/day) ⁺⁺	Excretion (lb/day) ⁺⁺			(% of intake)	(% of milk)
				Feces	Urine	Total	Total	Total
Petaluma/H	1.78	0.50	0.03	0.46	0.78	1.25	70	250
Petaluma/L	1.37	0.39	0.02	0.40	0.56	0.95	70	244
Modesto/H	1.29	0.39	-0.03	0.35	0.57	0.92	71	236
Modesto/L	1.19	0.28	0.05	0.34	0.53	0.86	72	307
Tulare/H	1.34	0.43	-0.06	0.37	0.59	0.97	72	226
Tulare/L	1.32	0.39	-0.01	0.37	0.58	0.95	72	244
Chino/H	1.49	0.33	0.09	0.40	0.67	1.07	72	324
Chino/L	1.57	0.27	0.13	0.41	0.75	1.17	74	433

⁺H and L corresponding to rations formulated for high and low performance strings, respectively.

⁺⁺1 lb/head/day equals 0.454 kg/head/day in SI unit.

Most herds have a consistent number of lactating and dry cows, over time, unless seasonal calving is practiced (which is very rare in California). Dry cows are fed a ration that has lower levels of most nutrients (e.g., N, metabolizable energy, fat, and most minerals) than that of lactating cows, and they eat much less of it. Arogo et al. (2001) estimated that a dry cow excreted 0.36 to 0.50 lb/head/day (163 to 227 g N/cow/day). There is no evidence that dry cows differ from the national average, and so the estimate of Arogo et al. (2001) is accepted.

Uncertainties about exact N excretion levels in dry cows have only a minor impact on dairy herd excretion estimates. Because dairy cows, on average, lactate for 305 days and are dry for 60 days per year, dry cows are a relatively small contributor to the total adult dairy herd N output on a commercial dairy. Hence, a 20% error in N excretion estimates from dry cows represents only a 1.5% error in N excretion estimates from all dry and lactating cows.

2.4 California Regional Differences that Effect Estimates of Nitrogen Excretion by Dairy Cows

N excretion by dairy cows in the various dairy production regions of California will vary due to regional differences in the milk production level of the cows, but there is no evidence that these differences are, or likely will ever be, of a sufficient magnitude to require additional design criteria for dairy farms, as long as expected manure and nutrient excretion are based on actual herd data (i.e., using methods 2 or 3 above to estimate manure nutrient excretion).

2.5 Role of Diet in Nitrogen Excretion by Dairy Cows

Traditionally dairy rations have been formulated based upon nutrients, rather than ingredients, to be both the lowest cost per unit of diet fed and to maximize milk production. The National Research Council Dairy Sub-Committee (NRC, 2001) provided recommendations on the nutritional requirements for dairy cows, and these are the nutritional basis used by virtually all consulting dairy nutritionists for formulating rations. While the utilization of dietary N by dairy cattle can be optimized by feeding ruminally degradable and undegradable N sources to NRC (2001) standards, there is no evidence that diet formulation principles, which are based upon nutrients, differ among regions of California that have cost effective access to different feedstuffs, and different 'region typical' diets.

The majority of California dairy herds (particularly the larger herds) utilize dairy cow diets that have been competently formulated to meet milk production goals. Typically, the whole farm N conversion efficiency from intake to milk is between 25 and 32% in California (Tables 2-1, 2-2). A recent survey in Merced County indicates that the range of whole farm N conversion efficiency in that county is between 20% and 34% (Alejandro Castillo, *personal communication*).

The feeding and management strategies of dairy cattle will impact nutrient levels in manure. For example, feeding cows close to the nutrient needs for their actual level of production will result in maximum N efficiency, and the least amount of N excreted per unit of milk produced. However in poorly managed dairies, cows may be fed poorly formulated diets and may not meet their production goals. In such cases, the N conversion efficiency from feed intake to milk produced will be reduced and N excretion per unit of milk production will increase proportionally (Table 2-2). At the same time, cows will eat less of the poorly formulated diet, thereby producing less urine and feces.

There are other strategies to improve feed utilization efficiencies on even the better-managed dairies. For example, administration of bovine growth hormone, extending photoperiod with artificial lights and milking three times daily have been shown to reduce nutrients in manure by 8, 5 and 6%, respectively (Kohn, 1999). N excretion of dairy cows can be minimized by appropriate management of the herd and, in this way, manure and the N excretion can be minimized to the limits of current knowledge.

2.6 Summary

The recommended average excretion of N by lactating dairy cows in California is 462 g N/head/day, and by dry cows it is 195 g/head/day. However the Committee stresses that these are average numbers, that the trend will be to higher values in the future as milk production per cow increases, that actual values will vary sharply among dairies, and that there are techniques available to make very accurate site specific estimates.

Short of measuring feed intake N, which allows for highly accurate estimates of manure N excretion using, e.g., the UC Dairy Animal Waste Model, the Committee believes that

expressing manure N excretion relative to milk production, rather than body weight or ‘animal units’, is the biologically most sensible assessor on within site efficiency of animal N use, as manure nutrient production is primarily a function of feed intake and feed intake is primarily a function of milk yield. There is no evidence that regional differences in California are of a sufficient magnitude to require additional region specific design criteria for dairy farms.

Dairies feeding better formulated diets will tend to have higher efficiencies (i.e., the lowest N excretion relative to milk production) while poorly managed dairies, where cows may be fed poorly formulated diets, will tend to have lower efficiencies (i.e., the highest N excretion relative to milk production). In contrast, the better managed dairies with higher per cow milk production will tend to have the highest manure N excretion per cow, thereby demonstrating the error in expressing manure N excretion on a per cow basis, rather than on the basis of milk produced.

There is no evidence that diet formulation principles that are based upon nutrients differ among regions of California, even if the cost effective access to different feedstuffs varies among regions. Hence, different ‘region typical’ diets need not be considered.

Chapter 3 - Distribution of Manure on Dairies

3.1 Introduction

The design elements and physical layout of a commercial dairy will determine where manure is deposited and how it will be collected. Three basic animal housing/manure collection categories are common in California: freestalls with flush systems, corrals with flush alleys, and corrals with no flush alleys². A small number of facilities collect manure from feed lanes via regular scraping or vacuuming. The feed apron is generally concrete, extends the length of the feed bunk, and ranges from 9 to 14' in width. Older facilities may have feed aprons as wide as 20'. Flushing frequency for corrals with flushed feed lanes or freestalls is variable and can occur multiple times per day or once weekly. Scraped or vacuumed systems are operated to relocate solid manure to storage or treatment areas or to open corrals. This can be done multiple times daily.

This chapter reviews our current knowledge about the distribution of manure within the various sections of a dairy (section 3.2), the partitioning of manure and nutrients between the solid and liquid phases of manure (section 3.3), and the partitioning into surface runoff (section 3.4).

3.2 Partitioning of manure on surfaces within the dairy

Dairy animals excrete feces and urine during their daily activities. It is generally assumed that manure is excreted based on residence time on each surface of the dairy. No published studies exist to describe manure production in each area on a dairy. Dr. Tom Shultz provided input into observations regarding manure accumulation at dairies (Shultz, 2000; *personal communication*). The overriding assumption was that animals excrete manure proportionally to the time they spend in various locations and that the excretion is uniform throughout the day for lactating cows. This assumption has been used and continues to be used by dairy design consultants who specialize in facility design, including manure storage systems (Harner, 2005; *personal communication*).

3.2.1 Feeding Areas

According to studies conducted in Tulare the average daily hours that cows spend on feed aprons varies by season, comfort modifications, ration form as well as the method and frequency of feeding (Shultz, 1989). Operators deliver feed from once to ten times daily and push up feed throughout the day. Some producers have environmental modifications at the feed apron, such as misters and fans, to reduce heat stress during hot weather. In general, cows remain on the feeding apron longer when these modifications are present. Total time for lactating cows on the feed apron ranges from 3 to 6 hours/day. The form of the feed can also affect time spent on the

² Freestalls are individual cow bedding areas where partitions orient the cow for comfort and sanitation (Stull et al., 1998). Manure deposited on the solid surface alley is collected daily (MWPS, 1985) (usually more than twice daily). Corrals are loose housing. Animal resting is not guided to any specific location. Feed bunk and water trough areas have scarified concrete areas. The concrete feed apron area can be cleaned via flush or scrape method.

feed apron. This may increase the estimated feeding time by as much as 0.5 to 1 hour/day for facilities that do not feed a total mixed ration.

Dry cows and replacement animals are housed in corrals and spend up to two to three hours a day on feed aprons. This amount of time can increase when the feed aprons are managed more intensively and/or inclement weather exists.

3.2.2 Milk Parlor

Animal housing and milking facilities are designed to minimize the time that cows spend away from corrals, generally 1 to 1.5 hours/milking. Manure deposited in sprinkler pens and milk parlors is removed by power spraying or flushing to a retention pond. Cows milked twice daily spend 2 to 3 hours in sprinkler pens and milking parlor. Cows milked 3 or more times daily may spend more time in this area. Facility design, traffic patterns, string size, and parlor throughput, define the time cows are away from their housing area (Smith et al., 2003).

3.2.3 Freestall Design

Design and maintenance of the freestalls affects the amount of time that cows spend in them. The micro-climate of the freestall will effect cow comfort and residence time in freestalls relative to those in adjacent corrals. Management of corral access varies, with some operators confining cows to freestalls when corrals are muddy or overly hot. Other producers impose no restrictions.

Properly designed and well managed freestalls will be occupied by the majority of cows, which may spend as much as 24 hours daily on concrete surfaces. Fans, misters and rubber matting in the feeding lanes are modifications that increase the time that cows spend in freestalls and on feed aprons. At facilities that do not have environmental modifications, cows may spend 3 to 6 hours per day feeding as part of the 8 to 16 hours per day spent in the freestalls. The remaining time is spent in the corral area.

3.3 Partitioning of manure nutrients into solid and liquid forms

Based on the observation of Shultz, the time that cows spend in various parts of the dairy can be generalized and manure collected as a liquid estimated (Table 3-1). The remainder of the manure would be collected in a solid form.

3.3.1 Liquid Forms

At dairies without flush concrete lane systems, only manure deposited in areas associated with milking (i.e., sprinkler pens, holding areas and milking parlor) is collected in liquid form (washwater) and enters retention ponds. For these systems, 8 to 19% of the total manure produced daily enters the liquid collection system (Table 3-1). Based on the total dissolved solids (TDS) concentration of dairy manure and based on the TDS recovered in wash water, an estimated 11% of manure produced in a dry lot dairy is in the liquid waste stream (Chang et al., 1974). When lactating cows are housed in dry lots where the feed lanes are cleaned by flushing,

an estimated 21 to 48% of the total manure produced enters the liquid stream, accounting for manure deposited during the milking and on feed aprons. In freestall dairies, 42 to 100% of the manure may be collected in the liquid stream. Management at a given facility may be such that 100% of manure is collected as liquid during some months and markedly less manure is collected as liquid in other months.

Bedding material is used to produce dry and comfortable freestall surfaces for cows to rest on. The sources of the bedding material vary. Coarse solids separated from liquid dairy manure, dried corral solids, almond shells, rice hulls, sawdust and straw have all been used, separately or in combination. The quantity of bedding material varies among dairies. This bedding will be transferred from the freestalls to the adjacent lane by cow activities. Bedding contributes to total solids and N loads of the dairy waste stream. In one freestall dairy, bedding was quantified to be 26 lb/cow/day (11.8 kg/head/day) (Meyer et al., 2004). Thus for a freestall flush dairy, it is reasonable to suggest that all of the manure excreted by the cows, plus the daily replacement bedding material, can be collected in a liquid form.

Table 3-1: Summary of estimated residence time for milking cows on concrete surfaces and estimated percent of manure collected as a liquid¹.

Type of Dairy	Estimated Time on Pavement (hr/day)				Liquid manure (%)
	Feeding	Milking	Stall/Alley	Total	
Corral w/o Flush	3 – 7	2 – 4.5	-	2 – 4.5	8-19
Corral + Flush	3 – 7	2 – 4.5	-	5 – 11.5	21-48
Freestall		2 – 4.5	8 – 22 (includes feeding)	10 - 24	42-100

¹It is assumed that manure is deposited based on residence time on each surface and location.

3.3.2 Solid Forms

Manure not collected as a liquid or slurry is collected as a solid. Additionally, solids can be removed from the liquid stream. Solid separation occurs primarily through mechanical or gravity separation (including weeping walls, screens, conveyor scraper) or by settling basins. Van Horn et al. (1998), in a study of Florida dairies, indicated that the stationary mechanical separators typically remove 20 to 30% of the organic matter from the liquid waste stream.

In California, mechanical and gravity separation systems have been evaluated (Meyer et al., 2003). Mechanical screens measured in California have removed 5% to 15% of total solids (TS) in a dilute (< 2%) TS stream. The efficiency of solid removal in gravity settling basins, measured at weekly intervals, ranged from less than 25% to more than 65% (*ibid.*). Short-circuiting (basin outlet near basin inlet) appeared to negatively affect the performance due to very short residence times. As basins accumulated sediments, more water found a short exit path and circumvented the basin's ability to reduce the flow velocity and settle out more suspended solids. Weeping walls consistently removed an average of more than 50% of the total solids (Meyer et al., 2004).

3.3.3 Salt and Nutrient Removal in Solid Separation Systems

Typically, most of the minerals (salts) and N remain in the liquid stream because they are either dissolved in the liquid or associated with the finest particle sizes (Figures 3-1 and 3-2, Table 3-2). Fine particles are not removed from the liquid stream by solid separation (Meyer et al., 2004). Van Horn et al. (1998) estimated that 5 to 20% of the N may be removed by screening when the mechanical solid separator is used in dairies in Florida. Chang and Rible (1975) size classified samples of freshly collected dairy manure. Their data also showed that the N contents of the waste material were size classified, with the highest N content found in the smallest size fraction (Table 3-2). More than 50% of the N in the liquefied dairy manure was in the <0.053 mm size fraction and would not be removed unless the solid separation process was effective to remove the very fine particles. Mechanical solid separators typically remove particles that are > 0.5 - 1 mm in size. Chang and Rible's data suggested that 25 to 30% of the N could be expected to separate in the process.

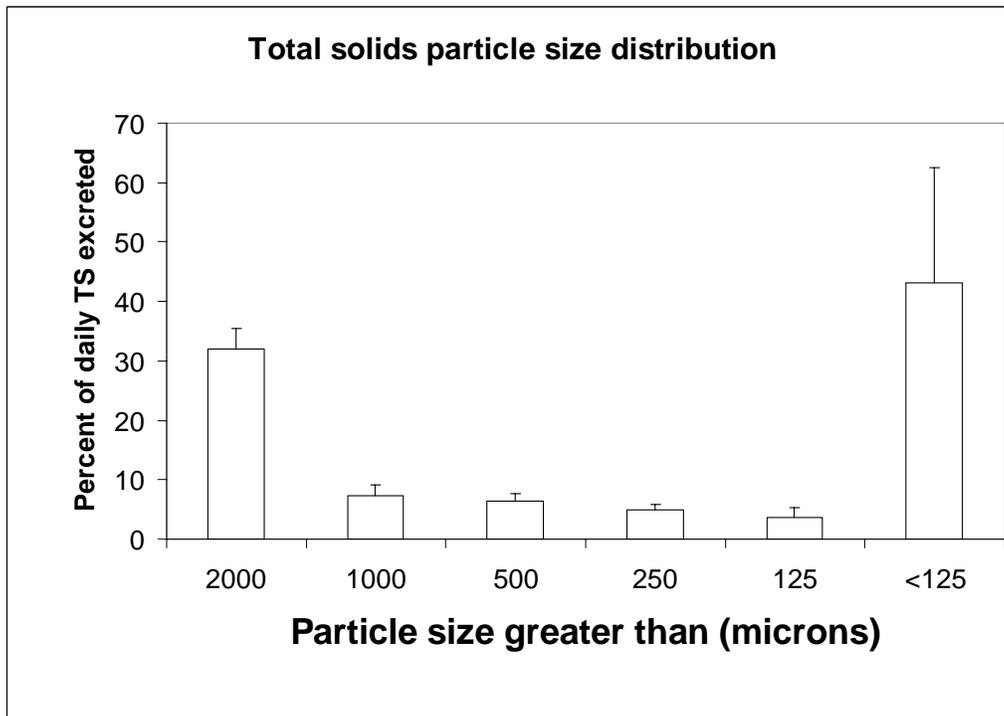


Figure 3-1: Particle size distribution in the feces and urine of four milking cows that were fed alfalfa based total mixed rations. Fecal dry matter can represent 88% to 95% of daily dry matter.

However, in a recent California study it was found that the N and macro-element nutrient content (salts) of influent and effluent samples did not significantly differ in a gravity flow separation system where approximately 50% of total solids were removed (Meyer et al., 2004). Soluble N and salts, in addition to fine and very fine suspended particles remained in solution (in the liquid manure) and were removed with solids only to the extent that water was removed in solids. For nutrient management purposes, documentation of mass of removal and nutrient content is

necessary to obtain site-specific nutrient removal credits. Without such site-specific measurements, nutrient removal estimates based on total solids removal are inherently unreliable and should not be used.

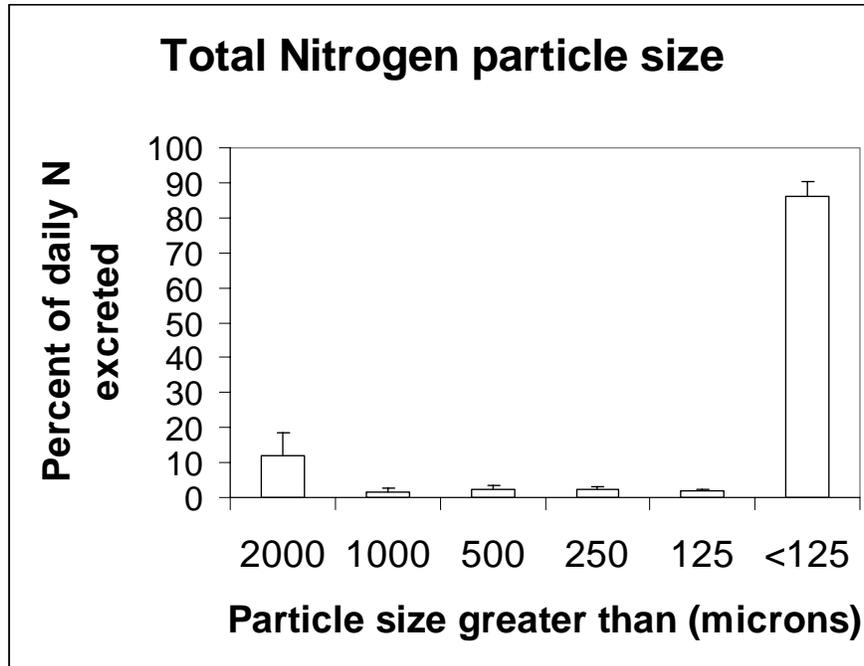


Figure 3-2: Nitrogen distribution among particle sizes in four milking cows that were fed alfalfa based total mixed rations. Most of the nitrogen in fecal matter is associated with the smallest particle sizes. Note that fecal N represents only 35% to 50% of the total daily N excretion (the remainder being in urine). Shown here are results for total manure (feces + urine).

Table 3-2: Particle Size Distribution of Freshly Collected Dairy Manure

Particle Size (mm)	N (% Dry Weight)	Mass (% Dry Weight)	N (% of Total)
<0.053	4.6	38	59.2
0.053 – 0.105	3.8	2	2.6
0.105 – 0.250	2.8	4	3.8
0.250 – 0.500	1.7	7	4.1
0.500 – 1.000	2.0	7	4.7
>1.000	1.8	42	25.6

Derived from data in Chang and Rible (1975)

3.4 Surface runoff

There is no method available to estimate nutrient runoff from corral surfaces. Runoff coefficients used by the Natural Resources Conservation Service (NRCS) are small for dairies in the San Joaquin Valley. Following the method in the NRCS Agricultural Waste Management Field Office Technical Guide liquid storage needs can be estimated. Monthly rainfall and runoff coefficients were multiplied and then summed across the winter months to estimate potential

runoff in the region between Lodi and Bakersfield, suggesting typical winter runoff ranging from 2.5 to 8 inches (December through March). This allows one to estimate runoff volume. However the nutrient content of the associated runoff is unknown. Standard practices throughout California include removal of manure from corrals prior to the rainy season.

Van Horn et al. (1998) estimated that approximately 1% of the excreted N on Florida dairies may be lost through surface runoff. The frequency, intensity and amount of precipitation in the major dairy regions in California are considerably lower than in Florida. The magnitude of N and salt runoff to liquid retention systems is therefore assumed to be negligible.

3.5 Summary

Manure distribution patterns vary depending on the facility infrastructure and operational and managerial decisions. Most likely, the manure distribution pattern will not be the same in any two dairies. Even within the same dairy, the deposition pattern shifts as operational parameters change. The amount of manure collected in liquid will vary from 8% (only manure excreted in the milking parlor) to 100% (manure from animals always living on concrete). More intensively managed systems (freestalls) will collect 42 to 100% of manure on a daily, monthly, or yearly basis.

There are no specific data to quantify the nutrients carried from the corral to the retention pond as a result of rain runoff. Data from Florida suggest this number is minimal.

Solids removal from mechanical and gravity flow separation systems are quite poor (under 25%) to good (consistently 50%). However, nutrient removal cannot be described based on a percent of solids removed. Soluble nutrients and salts predominantly remain in the liquid system.

Chapter 4 – Atmospheric N Losses from Liquid Dairy Manure prior to Land Application

4.1 Introduction

Atmospheric nitrogen losses from dairy manure can be significant and their quantification is important to improve management of N application to crops at agronomic rates. Atmospheric nitrogen losses occur as soon as feces and urine are mixed upon excretion, and continue throughout the material handling processes of collection, storage, treatment, distribution and utilization. The magnitude of atmospheric N loss from each process differs depending on the chemical and physical composition of manure and is influenced by environmental variables. Atmospheric N losses are primarily due to volatilization of ammonia, but also include smaller amounts of nitrous oxide and nitrogen gas. The Committee investigated ranges of atmospheric ammonium nitrogen volatilization specifically from liquid dairy manure in freestalls, flushlanes, and storage lagoons (“animal production area”). The chapter is divided into two sections: The first section (section 4.2) provides a discussion of methods to measure site-specific atmospheric N losses. The second section (section 4.3) applies the recommended approach to California dairies based on data collected by members of the committee. This chapter does *not* cover atmospheric losses that occur during or after land application of manure. These latter type of losses are discussed in chapter 5.

4.2 Estimating atmospheric N losses at existing dairies.

The National Research Council convened an Ad Hoc Committee on Air Emission from Animal Feeding Operations (AFOs) to review and evaluate the scientific basis for estimating the emissions from existing operations (NRC, 2003). The current approach of estimating air emissions using “emission factors” is based on measuring emissions from several facilities to obtain an average emission per production unit. A critical requirement for estimating appropriate emission factors is a statistically representative survey of emissions from a class of AFOs over several iterations of the time period to be represented. The NRC (2003) committee concluded that the existing emission factors for AFOs are generally inadequate and inappropriate because of the limited number of measurements on which they are based, as well as wide emission variability among AFOs. The NRC (2003) committee also concluded that it is impractical to individually assess atmospheric nitrogen (N) losses in the animal production area via direct measurement of volatilization on existing operations, except for research purposes.

The NRC (2003) report recommended replacing the currently used “emission factor” approach with process-based modeling to estimate site specific emissions. Process-based modeling involves analysis of the farm enterprise through a mass-balance constrained study of the physico-chemical nitrogen dynamics and mass transfers among its component parts (see, for example, Appendix B). The amount of N lost from one farm component affects the amount that can be lost from subsequent components. For example, if ammonia (NH₃) volatilizes from the freestalls, it cannot volatilize again from the waste storage lagoon unless additional ammonia has become available through mineralization of organic N. Moreover, transformations that occur in one of

the farm components might affect emissions and further transformations in other components. Process-based modeling uses nutritional mass-balance calculations for excretion estimation and physico-chemical, dynamic process representation to represent mass flows and losses of major elements (e.g., N, carbon [C] and sulfur [S]) within the animal production area as well as in the land application (crop production) area. To assess atmospheric N losses from existing dairies, site-specific process-based models should be used in conjunction with mass balance plans as suggested by the NRC (2003) recommendations.

Whole-farm and individual farm component mass balance estimates provide aggregated estimates of all N losses including atmospheric losses, runoff, and leaching. An N balance (i.e., farm imports and exports) for each farm component can be calculated and predicted by using software described by Dou et al. (1996) as recommended by NRC (2003). The N balance is computed from information about the total amount and N contents in imported crop and feed and in exported milk and animals. Farm-component specific N balances can be obtained by considering milk production and N excretion (see chapter 2), N distribution among farm components (see chapter 3), and N application for crop production (see chapter 5). The difference between imports and exports (i.e., N balance) in each component provides a site-specific upper limit for the maximum potential atmospheric N emissions to the environment (the loss term also includes runoff and leaching losses). Some of the records needed for such mass balance analyses are normally maintained by dairies for other (i.e., tax and management) reasons. However, for proper estimation numerous records of export/import quantities of N containing materials (i.e., animals, milk, feeds, crops) as well as their actual N contents are required and are not commonly collected on California dairies.

Actual atmospheric N losses are difficult to obtain from mass balance analyses alone as the mass balance analysis does not distinguish atmospheric, runoff, and leaching N losses within the various farm components. Hence, the mass balance approach can only be used as an upper limit check for process-based models that estimate site-specific atmospheric N losses.

4.3 Potential atmospheric N losses from liquid manure in California dairies

4.3.1 Approximate bounds of atmospheric N losses from the animal production area

N to P Ratio Method: On freestall dairies, estimates of net atmospheric N losses in the animal production area can be obtained by measuring changes in the nitrogen to phosphorus ratio: Unlike nitrogen, phosphorus (P) excreted by dairy cows is not lost via volatilization. Only small amounts of both nutrients are separated out into the solid fraction during solid-liquid separation (Meyer et al., 2004). Furthermore, wash water added to the liquid manure stream from the milking barn contributes relatively small amounts of either nutrient to the liquid manure stream and does not affect the N to P ratio. Hence, the only major process affecting the N to P ratio in the liquid manure stream between the time of excretion and the storage lagoon outlet is N volatilization. Also, where significant bedding materials (straw, composted manure, etc., see chapter 3) are imported to the dairy, the N to P ratio of the bedding material collected into the liquid manure stream may affect the overall N to P ratio in the liquid manure stream. In freestall dairies where bedding materials do not contribute significantly to the liquid manure stream

nutrients, comparison of the N to P ratio in freshly excreted manure (i.e., urine and feces) with the N to P ratio in manure collected from the lagoon yields an estimate of the overall atmospheric N losses from the production area (freestall lanes, flush system, and lagoon). We measured and compared N to P ratios in excreta and lagoons on 20 dairies in the Merced area (Robinson and Mitloehner, *unpublished data*). At the investigated dairies, the range of estimated atmospheric N losses, based on the N to P ratio reduction, was found to be between 20 and 35%.

Readily hydrolyzable organic N Method: Following a recommendation by the NRC (2003, p.109), an upper limit for atmospheric N losses that occur before lagoon storage can also be obtained by using the following simplified approach: The N excretion estimates in Chapter 2 of this report indicated that the average N excretion of dairy cows in California is 169 kg/head/yr. Approximately 50% (84 kg/head/yr) of total N is excreted in urine and 50% is excreted in feces. Approximately 70% (~60 kg) of the N in urine is in the form of urea, which is the N form that is readily hydrolyzed to ammonia and which is subject to volatilization. While some of the organic N from feces may be mineralized and therefore also be subject to volatilization, this process predominantly occurs in dry lot corrals and/or within lagoons rather than on flushed surfaces because it requires a relatively long process time. Based on these California-specific approximations and because of the relatively short residence time of excreta on these surfaces, the biological ceiling for atmospheric N losses from freestalls and flushlanes is approximately 35% of N excreted. This number is consistent with the above results from the N to P ratio measurements, although the latter accounted for losses not only in the freestall and flushlanes but also in the lagoon.

4.3.2 Atmospheric N losses from liquid manure storage

A process-based mechanistic model, as recommended by NRC (2003), can be used to quantify the dynamic interaction between the various factors that contribute to volatilization of ammonia from stored liquid dairy manure (e.g., lagoons). We developed an ammonia mass transfer model (see Appendix B for details on computation and validation) and calculated N volatilization from liquid manure storage lagoons under different conditions of lagoon depth and pH. Sample computations of ammonia volatilization from the surface of a waste storage lagoon were obtained for a lagoon with a volume of 9 million gallons, which is assumed to hold the liquid manure from 1000 cows for 90 days, at a rate of 100 gallons per cow per day. Two different depths, three pH levels, and NH₃-N concentrations were assumed for the lagoon based on the levels encountered on dairies. Ammonia emission rates are higher in warm weather than in cold weather. Over 65% of annual ammonia emissions occur in the five months from May through September. Model results for a dairy waste water lagoon under climate conditions representative of Fresno County are summarized in Table 4-1:

Table 4-1: Nitrogen Emission Calculations from Lagoons

Lagoon Conditions			NH ₃ -N Emission (kg head ⁻¹ yr ⁻¹) ⁺
Depth, m (ft)	pH	NH ₃ -N, mg l ⁻¹	
7.62 (25)	7.0	300	3.0 (6.6)
		450	4.4 (9.7)
		600	5.9 (12.9)
	7.4	300	7.4 (16.3)
		450	11.1 (24.4)
		600	14.8 (32.6)
	7.8	300	18.0 (39.6)
		450	27.3 (60.1)
		600	36.4 (80.2)
3.05 (10)	7.0	300	5.1 (11.2)
		450	7.6 (16.7)
		600	10.2 (22.5)
	7.4	300	12.7 (27.9)
		450	19.0 (41.9)
		600	25.3 (55.7)
	7.8	300	31.2 (68.7)
		450	46.8 (103.1)
		600	62.4 (137.4)

⁺Values in the parenthesis denotes NH₃-N emission in pounds per head of animal per year.

The calculations show a wide range of possible ammonia emissions from lagoons, ranging from 3.0 to 62.4 kg/cow/yr depending on lagoon depth and pH, for a specific location. The calculated average for ammonia losses from lagoons, based upon this research, is 19.4 kg/cow/yr. Using an average N excretion for dairy cows in California of 169 kg/head/yr (Chapter 2), we suggest that the range of atmospheric N losses from dairy storage lagoons is between 1.8 and 37%, with an average of 12%. Again, this range is consistent with the N to P ratio estimates and the maximum hydrolysable N estimates.

Table 4-2: Estimated atmospheric N Loss (kg N/head/yr) from dairy manure in the production area of a typical freestall dairy in the SJV assuming 169 kg N excreted/cow/yr.

	Liquid manure N-losses, Production Area	
	Range, %	Average, %
Freestall + Flushlanes + Lagoon ¹	20-35	30
Freestall + Flushlanes, Maximum ²	35	35
Lagoon ³	2-37	12

¹ based on measured N to P ratio, ² maximum hydrolyzable N method, ³ process-based model (Appendix B)

4.3.3 Atmospheric N losses: Dependency on frequency of freestall flushing, on use of lagoon recycled (vs. fresh) water for flushing and on time spent in retention ponds

More frequent flushing of freestalls will certainly decrease atmospheric N losses from freestalls and flushlanes as it removes the highly reactive urine urea and fecal urease to the pond, where depth and pH conditions will sharply curtail urea N volatilization as ammonia. In addition, the incremental effect of more frequent flushing on suppressing ammonia volatilization will be less with each subsequent increase in flushing frequency.

In the flush lanes, use of recycled lagoon water for flushing will expose lagoon water urea to a very shallow water depth, agitation, and to urease from fresh feces. These circumstances typically lead to increased ammonia volatilization. But the amount of volatilization from recycled urea in the flush lane is currently unknown and may be quantitatively trivial.

Nitrogen emissions from storage ponds (lagoons) will increase with time spent in the lagoon. However the amounts volatilized will vary dramatically due to factors such as lagoon depth (which impacts its surface area), pH, ambient temperature and wind speed over the surface.

Overall there are insufficient data to quantify atmospheric N losses associated with effects of frequency of freestall flushing, use of lagoon recycled (vs. fresh) water for flushing and time spent in retention ponds.

4.3.4 Total atmospheric N losses in the animal production area

Considering the above estimates of potential atmospheric nitrogen losses from freestalls, flushlanes, and lagoons (Table 4-2), we suggest that the atmospheric N losses in liquid manure from typical dairies in the Central Valley ranges from approximately 20% to approximately 40% of excreted N. We emphasize that these are approximate *average* losses across multiple dairy farms and that these do not include atmospheric losses during the land application of manure. Given the above estimates, actual losses on individual dairy farms may vary widely prior to the land application process, but are unlikely to exceed 40%. The wide range partly stems from differences in manure distribution patterns within and among dairies (Chapter 2) and from differences in residence times of the animals on flushed concrete floors. Also recall that on dairies with flush systems, the fraction of manure collected into liquid storage may range anywhere from 21% to 100% (Table 3-1). Not all manure N reaches the liquid storage structure, particularly if solid-liquid separation is in use. Furthermore, highly variable amounts of excreted manure can be deposited on non-flushed surfaces.

4.4 Summary

Use of a universal animal ‘emission factor’ for reactive N compounds (e.g., ammonia) from commercial dairies is not possible because of the limited number of field measurements on which they are based and the wide emission variability among and within dairies. We concur with the National Research Council (NRC, 2003), that there is no single emission factor that could possibly describe atmospheric N losses from dairies.

To determine atmospheric N losses from existing dairies, we concur with NRC (2003) in recommending the use of process-based models coupled with a whole-farm and farm component N balance approach that describes potential atmospheric N losses for the different farm component processes. The process-based model approach coupled with total N balance predicts atmospheric N losses between excretion and land application and helps manage nutrient application rates to crops at agronomic rates. The Committee emphasizes however, that while this approach is technically viable, it requires extensive data measurement, record keeping, and is associated with significant estimation errors.

There are insufficient data available to quantify atmospheric N losses associated with effects of frequency of flushing, use of recycled water for flushing and time spent in retention ponds. However, in general, more frequent flushing, of fresh water for flushing and shorter residence times in lagoons will tend to decrease ammonia volatilization. However the quantitative impact of these strategies is unknown at this time.

In light of these findings and in light of California-specific conditions, we suggest that atmospheric N losses from liquid manure (i.e., freestalls and flush lanes and lagoons) used for dairy planning and permitting purposes, are considered to range between 20% and 40%. The use of a single number (“emission factor”) is strongly discouraged. Note, that these losses do not include atmospheric N losses in the land application (crop production) area, which are discussed in chapter 5.

Chapter 5 - Developing Nitrogen Application Rate Guidelines for Crops Receiving Dairy Manure

5.1 Dairy Farming and the Nitrogen Cycle in Crop Fields

Green plants use nitrogen (N) to manufacture proteins, chlorophyll, and other essential plant biochemicals necessary for their growth. Plants acquire N primarily from soils within the root-zone. Most of the N in soil is a part of the soil organic matter. For prevention of a long-term decline in the soil organic matter, N must be added at least at rates that will replace the N removed in the harvested crop. But terrestrial ecosystems, agricultural or natural, are inherently inefficient, and leaching of N beyond the root extraction zone is unavoidable; N must also be added to the soil to compensate for non-harvest, non-leaching system losses (Fig. 5-1).

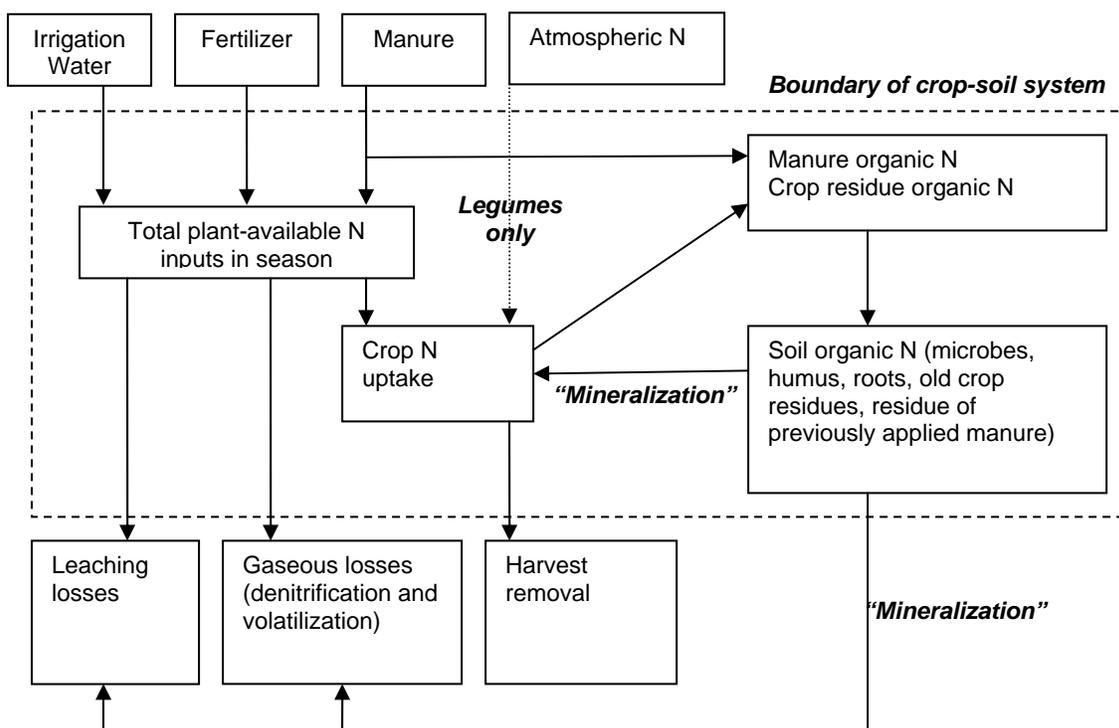


Fig. 5-1: Major Components of the Nitrogen Cycle in a Forage Crop Fertilized with Dairy Manure and Commercial Fertilizer.

A thorough understanding of N sources (inputs) and sinks (output and losses) in the soil root-zone (“soil”³) and the dynamic interaction between N sources, the crop-soil system, and the N

³ Note that this review does not address losses in the deep vadose zone above the water table but below the root-zone. For purposes of this document, the term “soil” refers only to the uppermost 3 ft – 6 ft (root zone) of the unsaturated zone.

sinks will provide relevant information to assess how much fertilizer or manure N must be applied to meet the crop N uptake demands and to compensate for harvest N removal and N losses from the crop-soil system. An assessment of the soil nitrogen cycle provides the basis for determining agronomic rates of N application.

The dynamics of nitrogen within the crop-soil system are more complex than those of other nutrients such as potassium (K). This is due to the fact that nitrogen within the crop-soil system occurs in three different forms -- organic nitrogen (N_{org}), ammonium (NH_4^+), and nitrate (NO_3^-). Almost all organic nitrogen from crop residue, in the soil organic matter pool, and in manure applications is immobile within the soil, strongly sorbs to soil particles, and is not available for plant uptake. Mineralization of organic nitrogen to ammonium is a microbial process that depends on the availability of carbon as a microbial food source and favorable temperature, moisture and other growth conditions. Ammonium nitrogen is plant-available, but also sorbs to soil particles. Under most California conditions, ammonium nitrogen is rapidly (days to weeks) converted to nitrate (nitrification). Nitrate is plant-available, is not adsorbed to soil particles, and readily moves with soil water.

Agronomic application rates must be defined within the constraints of the nitrogen cycle and its dynamics within the crop-soil system. The need to satisfy nitrogen and other nutrient uptake by crops is shared by all agronomic systems. Cropping systems on dairies in the Central Valley, however, differ from most other agronomic systems in two significant ways:

- The use of manure instead of or in addition to commercial fertilizer (similar only to organic farming systems).
- High nitrogen throughput due to production of high-biomass crop species, and two crops per year on much of the acreage. (See for example Table 5-1).

Table 5-1: *Examples of observed dairy forage field N inputs and harvest removals.*

Dairy	Cropping	Conventional ²		Improved ³	
		Input (lbs/A)	Removal (lbs/A)	Input (lbs/A)	Removal (lbs/A)
1	Wheat + Corn	1429	872	1657	844
2	Corn	1076	418	1000	518
4	Triticale + Corn	1670	758	508	740
5	Corn	1154		1032	
6	Corn	714	326	588	379
7	Corn	530	458	504	479
8	Wheat + Corn	1022	619	1232	680
10	Corn	1601		1277	
11	Corn	718	558	806	503

¹Data from Appendix D, "Integrating forage production with dairy manure management in the San Joaquin Valley", Sept. 2001, Annual Report (Year 2) to UC Sustainable Agriculture Research & Education Program, Davis, CA.

²Conventional fertilization practice with N from chemical fertilizer and dairy manure.

³N fertilizer from dairy manure only.

In considering the appropriate agronomic rate for nutrient management with manure, it is important to understand that manure-based nutrient management further differs from standard commercial fertilizer-based nutrient management systems in three important aspects:

- the application method is different
- manure is a composite (multi-nutrient) fertilizer
- a significant amount of the nitrogen applied is in the organic form

The first issue (manure application method) is rapidly evolving as technologies are currently developed to properly deliver and mix liquid manure with irrigation water that is applied to the crop. This issue is beyond the scope of this report. The second issue, manure composition, is discussed in chapter 2 of this report. The third issue, application of significant amounts of organic N in manured systems, speaks to the fact that much of the applied nitrogen must be mineralized before it becomes plant-available. Knowledge of nitrogen mineralization rates is therefore critical to proper management of nutrients in a manure-based system.

In this chapter, we will review the main components of the N cycle relevant to N management at Central Valley dairies where manure is applied to irrigated forage crops. On the source (input) side, these components include commercial fertilizer, manure, atmospheric deposition, and irrigation water nitrogen (section 5.2). On the sink (output and loss) side of the N cycle, the key components are harvest removal (section 5.3), gaseous losses, and leaching losses (section 5.4). Mineralization of the organic N (section 5.5) exerts significant control on the soil N dynamics and on the sink pathways (section 5.6) and is therefore given attention here. We use the information to discuss appropriate agronomic rates for California dairy farms (section 5.7).

5.2 Sources of Nitrogen (Nitrogen Inputs)

The primary sources of N for non-leguminous crops grown on California dairies are manufactured fertilizers and animal manure. Soil amendments may also contain significant amounts of N. Additional N is present in some irrigation waters as nitrate, and this may be a significant source of nitrogen (Table 5-2).

Table 5-2: Total nitrogen input [in lbs N/A] from irrigation water application as a function of the nitrate-N concentration in the irrigation water (left column) and of the amount of irrigation water applied (top row). In metric units, 1 mg L⁻¹ nitrate-N in 10 cm irrigation water is equivalent to 1 kg N/ha.

Nitrate-N [ppm or mg L ⁻¹]	-----Irrigation Water Application [acre-inches/acre]-----						
	4	8	12	18	24	30	36
4	4	7	11	16	22	27	33
7	6	13	19	29	38	48	57
10	9	18	27	41	54	68	82
15	14	27	41	61	82	102	122
20	18	36	54	82	109	136	163
30	27	54	82	122	163	204	245
40	36	73	109	163	218	272	326

Atmospheric deposition of nitrogen oxides and ammonia are another source of N to cropland soils although the total annual deposition is relatively small. Total wet and dry N deposition in

the San Joaquin Valley is approximately 14 lbs/A/year (16 kg N/ha/year) (Blanchard and Tonnessen, 1993; Mutters, 1995).

Atmospheric nitrogen also enters the crop-soil system via biological nitrogen fixation (BNF). In Central Valley dairies, the forage crop rotations commonly include legumes, mostly alfalfa and some clovers. Bacteria living in nodules in association with roots of these crops fix atmospheric di-nitrogen (N_2), and therefore leguminous crops do not depend on other external sources for their N requirements. However, if N is available in the root zone, these crops will preferentially utilize the available soil N, and the microbes will only fix the atmospheric N to supplement their needs. Alfalfa is well known as an efficient scavenger of N and other nutrients from the soil. If it is well managed, alfalfa during the second and third years of production typically removes 400 to 500 lb N per acre per year in the harvested crop (Table 4-1, CPHA, 2002). Alfalfa N uptake from soil proportionally reduces the amount of biologically fixed N.

What is the net gain in soil N resulting from a typical three-year stand of alfalfa? It is well established that non-leguminous crops following alfalfa invariably need less N fertilizer than when following a non-legume crop. This reduction in N fertilizer need is referred to as a “legume credit.” The legume credit does not refer to the amount of biologically fixed N, but rather the net reduction in fertilizer N requirement to the following crop that is due to the legume in the crop rotation. The legume credit is estimated to be 50 to 80 lbs of N per acre, depending on how strong the alfalfa production was in its final year⁴.

5.3 Crop N Uptake and Harvest Removal

5.3.1 Seasonal N Uptake Total

Plants can take up only inorganic forms of nitrogen, the main forms being NO_3^- and NH_4^+ . These forms are referred to as available nitrogen. Plant nitrogen uptake has two important components: (1) The total nitrogen uptake of the crop and (2) the portion of the uptake that is eventually removed from the field at harvest. The total plant nitrogen uptake provides a target to assess the nutrient status in the soil relative to the crop demand. Plant nitrogen uptake needs must be available in the soil to ensure a crop that produces an economically acceptable yield and quality.

The second component, crop removal, is important since that is the minimum amount that must be replaced with fertilizer (chemical or manure or from other external sources) if the soil organic N reservoir is to be maintained. The portion of the plant that is not removed by harvest (roots, stubbles, etc.) will decay in the field and its nitrogen is returned to the soil nitrogen pool. That portion of the total plant nitrogen uptake therefore does not need to be replaced with external N inputs.

For long-term fertilizer management, the second component provides the application targets. Approximate N contents per unit of crop yield are published in Table 4-1 of the Western Fertilizer Handbook (WFH), 9th edition (CPHA, 2002). That table presents crop N removal in

⁴ E-mail survey of UC Cooperative Extension farm advisors and specialists conducted January 2005.

amounts per acre per unit of crop yield. If actual field harvest histories are not available, these data (both 9th and 8th edition) can be useful to estimate the crop removal of N from the soil, with some precautions. WFH values are preferable over historic data when historic data reflect harvest from excessive nutrient applications. These were likely subject to luxury N uptake that will not be maintained under balanced nutrient management.

Some important forage species produced by Central Valley dairies are missing from Table 4-1 of the WFH, notably cool-season grass forages such as wheat, triticale, oats, and annual ryegrass, which are most commonly harvested by dairy producers for silage or green chop, rather than for grain or hay. For convenience, these are referred to here as “winter forages”.

Nitrogen uptake by winter forage crops (here reported in lbs N per ton at 70% moisture) varies by large amounts due to varying cutting stage and due to varying soil nitrogen conditions between individual fields, between individual farms, and from year-to-year. In California, under reasonably fertilized conditions, the average nitrogen removal has been found to be 9.1 lbs N per ton to 10.5 lbs N per ton, typically harvested at the later milk to soft dough stage (Roland D. Meyer, *personal communication*; Peter Robinson, *personal communication*). In the Northern San Joaquin Valley, field research on winter forage nutrient uptake has been and is currently being conducted by UC Cooperative Extension (Campbell-Mathews, *personal communication*): Among 56 individual variety-location-years, representing a range of fertility conditions, nitrogen removal averaged 13.8 lbs of nitrogen per ton of forage when harvested at the boot or early heading stage, and 11.3 lbs N per ton when harvested at the milk to soft dough stage.

Overfertilization may increase the nitrogen uptake of winter forages. However, when nitrogen concentrations are much higher than the 10.5 lbs N per ton at the milk to soft dough stage, an unknown but potentially significant fraction of the N can be in form of nitrate, which may cause nitrate poisoning in cattle (particularly, if the cattle have not been adjusted to high nitrate feeds). Optimal fertilization balances the need to maximize crop growth and nutrient uptake with the need to limit the risk of nitrate toxicity in winter forages and the risk of nitrate leaching to groundwater. For planning winter forage plant N uptake, we recommend to use 10 lbs N per ton of forage at 70% moisture, harvested at the later milk to soft dough stage, as a preliminary guideline. Forages harvested at earlier growth stage (boot and flower) will have a higher per-ton requirement (but their harvested weight per acre is lower). Further research is needed to clarify optimal nitrogen removal in winter forages in California with respect to nutrient management (to ensure sufficient yield while avoiding excessive fertilization) and with respect to animal toxicity (high levels of nitrate in forage).

Multiplying these removal rates (lbs per ton) by the yield allows the removal to be calculated for a range of yields using the approach taken in Table 4-1 of the WFH (9th ed.). Additionally, 8.7 lbs K₂O and 3.6 lbs P₂O₅ per ton of forage (70% moisture content) are removed. Actual K and P uptake may vary greatly. In winter forages, K₂O removal of 18 lbs per ton or more are not uncommon and considerably higher P₂O₅ removal has been documented as well. Note, that levels above 18 lbs K / ton of forage may lead to severe animal health problems including death.

Some additional precautions in the use of Table 4-1 of the WFH are noted here:

- As stated in a footnote accompanying the table, “Actual nutrient removal may vary by 30% or more.”
- Moisture contents of the harvested materials are not given in the table, and the reader is left to assume that the nutrient concentrations displayed are at “standard” moisture contents i.e. grain and hay moisture contents are typically 10-12% while forages harvested for ensiling typically have a 70% moisture content.

To summarize, where actual data are not available, the WFH values (together with consideration of N system losses and all N inputs) can be used for assessing adequacy of crop land area for receiving manure. Where at all possible, such an assessment should rely on historical yields and plant N content representative of specific site conditions.

5.3.2 Timing of Nitrogen Uptake by Forage Crops Grown in the Central Valley

Dairy operators in the Central Valley typically grow silage corn during the summer and cereal forages during the winter.

The cumulative corn nitrogen uptake forms an S shaped pattern, with low uptake during the first 30 days of growth, then rapid uptake until silking. Uptake after silking is slower. Classic work conducted by Hanway (1962) indicated that 75-80% of nitrogen in corn is taken up by the time of silking. More recent research (Karlen et al. 1988, Schepers, unpublished data) indicated that modern silage hybrids with a “staygreen” trait continue to take up significant amounts of N after silking. Those results are in agreement with the results of on-farm studies over six location-years near Hilmar, California, in which only 64% of nitrogen (range 52-79%) was taken up by corn at the silking stage (Campbell-Mathews, 2001).

The cereal forages (also referred to as “winter forages”) are usually planted in mid- to late fall. Multiple-year field trials on N uptake by winter cereals have been conducted by Campbell-Mathews (2003 and 2004). The trials show that N uptake of winter cereals grown in the Central Valley also follow an “S” shaped cumulative N uptake curve, with relatively low rates of nitrogen uptake during December and January. By mid-January, only about 50 lbs N/acre has typically been taken up in the aerial plant portion from a mid- to late fall planting. The exponential growth phase starts in mid-February to mid-March, depending on the species and variety of cereal. For late fall plantings of winter forages, as much as 85% of the nitrogen uptake takes place during late February, March and April. During that growth stage, the rates of nitrogen uptake can be very high (Table 5-3).

TABLE 5-3: *Percent of total mid-April nitrogen uptake in N. San Joaquin Valley by period*

planting date	early maturing varieties			later maturing varieties		
	Mid-Oct	Early Nov	Late Nov/ Early Dec	Mid-Oct	Early Nov	Late Nov/ Early Dec
% thru Dec.	30	20	6	30	20	7
% January	14	12	7	12	14	8
% February	24	29	31	15	21	15
% March April	32	40	57	43	45	71

This period of rapid uptake can be exploited to allow application of higher rates of nitrogen in the usual single water-run application of dairy nutrient water than would be applied to a summer corn crop, where 40-60 lbs of N per acre is a typical target for a single application.

Sometimes, winter forages are planted earlier in the fall, from mid-September to mid-October. Since growing conditions are usually favorable during this period, nitrogen uptake will be considerably higher in the early fall than with later planting dates, with as much as 30% of the total nitrogen uptake occurring before January (Campbell-Mathews, 2003).

When the N uptake of summer silage and wheat are combined, the annual N uptake pattern by a corn silage and late-fall-planted winter forage rotation in the Central Valley follows the pattern shown in Figure 5-2. Most of the nitrogen uptake in this system occurs during relatively short periods in the spring and in mid-summer. The actual uptake timing depends on planting date, weather, crop variety/species, nutrient availability, and for the winter forage, whether there are multiple cuttings.

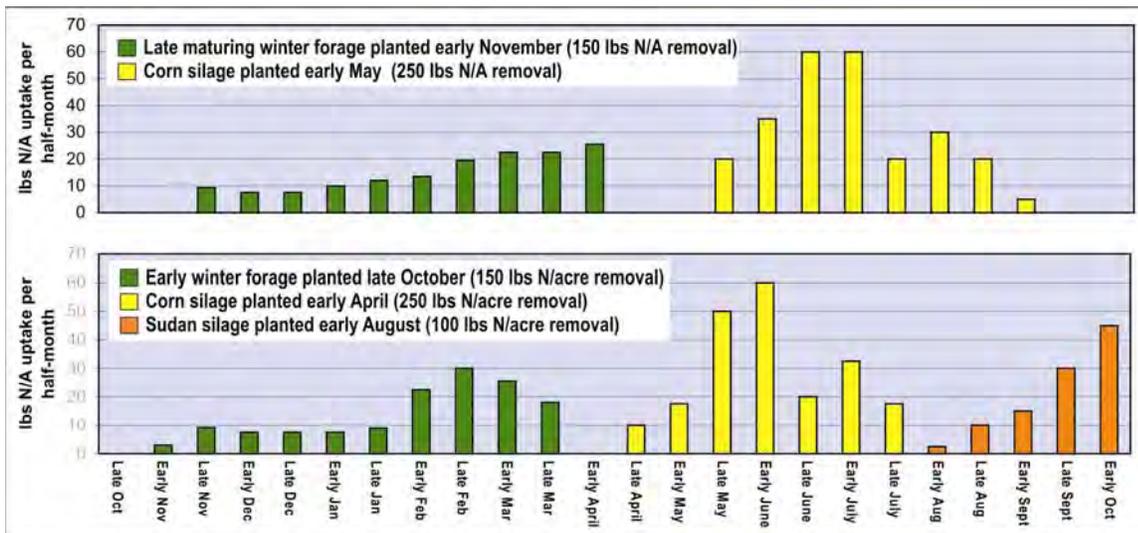


Fig. 5-2: Potential N uptake rate for a corn/winter forage double crop (top) and for a corn/sudan grass/winter forage triple crop (bottom) in the Central Valley of California.

5.4 Nitrogen Losses

In addition to the N uptake of the crop, there are five other pathways through which N may be removed from soils, namely:

- Ammonia volatilization from the soil surface
- Ammonia and other nitrogen gases volatilizing from plant surfaces
- Denitrification losses of nitrate and nitrite as N₂, N₂O, and NO gasses.
- Leaching of nitrate and nitrite beyond the root zone
- Surface runoff in solution or in sediment

The last pathway, surface runoff losses, will not be further discussed here. In California, by regulation, farmers must in as much as possible, prevent runoff from fields to which animal manure has been applied. The following review discusses the role of the remaining four pathways within the context of the N inventory in forage production soils amended with dairy manure.

5.4.1 Ammonia Volatilization from Soil

As manure is excreted from the animal, it is primarily organic nitrogen, but a portion of it is quickly converted to ammonium (NH_4^+). Under alkaline conditions, a portion of the ammonium N is in the gaseous ammonia (NH_3) form and subject to volatilization loss to the atmosphere (Bouldin et al., 1984).

We are not aware of research in which NH_3 losses have been measured during application of diluted lagoon water via surface gravity irrigation systems, by which method nearly all lagoon water is applied in California. In those systems, incorporation occurs through infiltration except that larger solids containing some of the organic N remain unincorporated on the soil surface.

A nearly 40-year-old fertilizer industry advisory states that up to 100 ppm (mg L^{-1}) $\text{NH}_3\text{-N}$ (22 lbs $\text{NH}_3\text{-N}$ per acre inch) may be safely applied in surface irrigation water before ammonia losses become significant (Warnock, 1966). It has therefore been suggested that the $\text{NH}_4\text{-N}$ content in irrigation water that is blended with manure water be kept near or below this level to avoid significant N volatilization. Manure water in the storage lagoon contains from 50 to 1,000 ppm (mg L^{-1}) $\text{NH}_4\text{-N}$ (Campbell-Mathews et al., 2001b), with typical concentrations of 200-500 ppm (mg L^{-1}). During irrigations, farmers commonly dilute lagoon water with 5 to 10 parts of fresh source water. This would result in 20 to 100 ppm (mg L^{-1}) $\text{NH}_4\text{-N}$ in the irrigation water. When the crop has full canopy coverage, the leaves will likely reabsorb some of the volatile NH_3 in the surface irrigation. Furthermore, the presence of the crop canopy reduces wind speed at the soil surface, reducing the upward flux of NH_3 . Temperature and irrigation water pH are also important controlling factors. Maximum NH_3 loss rates will occur with high wind, high soil pH, negligible crop canopy, high temperatures, and high NH_4^+ concentration in the irrigation water and in the soil. In our judgment, there is a potential for significant N loss only in undiluted or slightly diluted manure water applications (applications at the upper end of the $\text{NH}_4\text{-N}$ concentration range) particularly if applied during the early growth of the crop. In systems with frequent, but well diluted manure water applications, ammonia losses from the ground surface will commonly be minimal during the irrigation (10% or less).

5.4.2 Ammonia Losses from Plants

Several researchers have observed significant NH_3 loss directly from plants. Francis et al. (1993) analyzed the N fertilizer recovery by irrigated corn in which ^{15}N isotopic techniques were used to track the N distribution. The study was conducted at different sites, in different years, with fertilizer rates ranging from 45 to 270 lbs N ac^{-1} . The post-flowering N losses from the above-ground biomass of corn were quantified under the different N input regimes. When the isotopic

dilutions were factored in, the N losses from aboveground biomass ranged from 40 to 72 lbs N ac⁻¹, which amounted to 52 to 73% of the N unaccounted for in ¹⁵N balance calculations. While the mechanisms for the absorbed N to be lost from plant tissue are unknown, Francis et al. (1993) concluded that the observations were convincing and that failure to include direct plant N losses when calculating an N budget leads to overestimation of losses from the soil by denitrification, leaching, and ammonia volatilization directly from the soil.

At present, however, little data is available in the published literature to substantiate the reported observations. As the N balance is customarily conducted at harvest to account for the applied N, the N loss through the plant tissue becomes part of the overall losses (volatilization, denitrification, leaching) and does not change the perspective with respect to the maximum N requirements for plant growth.

5.4.3 Denitrification

Seasonal losses of N due to root zone denitrification are difficult to predict, as the rate of the process may vary dramatically over time and space. Ryden and Lund (1980) reported that total N losses due to denitrification in irrigated soils planted with vegetables ranged from 85 to 208 lbs N ac⁻¹ yr⁻¹. In another study, when liquid dairy manure was applied by sprinkler to a loamy sand soil for production of silage corn and bermudagrass/ryegrass hay, the estimated denitrification loss ranged from 11% to 37% of the total manure N applied (Lowrance et al., 1998).

Many irrigated dairy forage fields in the San Joaquin Valley are low in soil organic matter (<2%), are classified as excessively to moderately well drained, and receive regular applications of dairy manure. For that combination of characteristics, Meisinger and Randall (1991) in a review of the topic estimate that denitrification losses will vary from 8% to 28% of inorganic N in soils. In a typical double-cropped forage system in the Central Valley, the annual N inputs range from 350 to 675 lbs of N ac⁻¹. Based on the values of Meisinger and Randall (1991), the annual denitrification N loss from fields receiving dairy manure would range from 30 to 190 lbs N ac⁻¹.

On the other hand, for manure systems in a loamy sand soil in Merced County, Harter et al. (2001, 2002) and Campbell-Mathews (2004) concluded that the combined denitrification and volatilization losses were negligible (or within measurement error of leaching losses) compared to leaching losses to groundwater (Appendices G, H). Campbell-Mathews (2004), for example, reported total nitrogen losses (leaching, volatilization, and denitrification) of 159 lbs ac⁻¹ yr⁻¹ for the last five years. These losses resulted in recharge water nitrate-nitrogen (NO₃-N) concentrations of 25-50 ppm (mg L⁻¹), a concentration that is only possible if nearly all losses are attributed to leaching.

5.4.4 Leaching of Nitrate from the Root Zone

Leaching of water and solutes below the root zone can be minimized but not entirely eliminated, especially with the furrow and border check irrigation systems used by most dairy farmers in the Central Valley. In these gravity flow systems, the water distribution is susceptible to the variations of topography, soil texture, surface conditions, and soil moisture content across the

field. The irrigation schedule typically is a compromise to balance between irrigating the heavier, clay-rich portions of the field and the lighter, sandier portions of the field. Irrigating the heavier portions of the field too early would cause yield loss due to saturated soil conditions, while irrigating the more sandy portions of the field too late would cause crop water stress. Application amounts are usually determined not by how much water is needed to refill the soil profile in the root zone, but by how much water is needed to reach the end of the field. This may result in excess total water being applied and/or more water infiltrating the soil at the head of the field than in other parts. The non-uniformity in water application invariably leads to nitrate leaching past the root zone in some parts of the field if nitrate is available in the root zone. Furthermore, salts introduced through applications of dairy manure, fertilizer, and irrigation water will accumulate and concentrate in the root zone. They must be leached to sustain healthy plant growth. In the Central Valley, rainfall coupled with water losses due to non-uniform irrigation is often adequate to remove the excessive salts and maintain an annual balance. If leaching due to inefficient irrigation were entirely eliminated, salts would accumulate in the root zone to harmful levels during low-rainfall years.

In the Central Valley, the practically achievable irrigation application efficiency (IAE) for gravity flow irrigation systems is 70 to 85%, depending on the soil texture, size of the fields, and other field-specific factors (Hanson et al., 1999). This is equivalent to a leaching fraction (LF) of 0.15 to 0.30 in which LF denotes the fraction of total water input that is leached beyond the root zone. Under those conditions, nitrogen leaching is minimized by using split applications of fertilizer or manure (e.g., Harter et al., 2001; Campbell-Mathews, 2004; Nakamura et al., 2004). Simulations of a typically irrigated dairy forage crop system in the San Joaquin Valley indicate that N losses can be managed to be as small as 10% - 15% of the applied and available N, *after* subtracting denitrification and volatilization losses (Appendix C).

5.5 Mineralization of Organic Nitrogen

The rate of mineralization of organic nitrogen is dependent on microbial processes acting upon various organic compounds under favorable temperature, moisture, and other conditions. Mineralization of organic nitrogen in manure added to soil generally follows an exponential pattern, i.e., a very rapid initial rate followed by an increasingly slower rate as the more labile organic compounds are decomposed. The mineralization rate is often expressed in the form of a half-life of (labile) organic nitrogen (Appendix D). Half-life refers to the time need to mineralize half of the organic nitrogen applied. A longer half-life of the organic nitrogen corresponds to a slower mineralization rate. A review of field studies (Appendix D) indicates that typical nitrogen half-lives for organic N under California conditions likely range from a few tens of days to a few hundred days.

The associated dynamics of the organic nitrogen subject to various mineralization rates that vary seasonally according to a typical California climate were investigated by Guanglon et al. (Appendix E). They show that essentially complete mineralization of manure organic nitrogen (more than 95%) may be achieved within a 1 to 7 year time period depending on the mineralization rate, with organic N half-lives ranging from 50 to 280 days. Mineralization rates are higher during the summer (2-4 times faster than in winter and early spring), hence organic N

applied in spring and early summer will be mineralized much faster than organic N applied in the fall or winter. For mineralization rates corresponding to a half-life of 90 days or less, a long-term quasi-steady-state in the organic nitrogen pool is achieved within the second year after beginning manure applications, whereas it takes 5 years to achieve quasi-steady-state for a half-life of 280 days. Modeling analysis of soil water and nitrogen dynamics using data from a Merced County dairy field study indicate that, in sandy soils, mineralization rates appear to be relatively rapid (half-life of 90 days or less, Appendix H).

Across most of the likely range of mineralization rates, organic N can be shown to be a steady, relatively slow-release source of plant-available and leachable nitrogen. The steady release of mineral N from the organic N source is in contrast to the strong seasonal variability in plant uptake needs (Fig. 5-2). The mismatch between the rate at which organic nitrogen becomes plant-available nearly year-round and the rate at which nitrogen is seasonally needed by the crop leads to an inherent limitation in manure nutrient management: Applying manure with high organic nitrogen content may not provide sufficient nitrogen to meet the crop N demand during the most rapid growth stage, yet far exceed crop N uptake during the remainder of the cropping season (and during fallow periods), when it is potentially subject to leaching.

5.6 Long-Term Dynamics of the Soil Organic Nitrogen Pool

Where farmers rely heavily on organic N sources (as in organic farming or conventional farming with livestock manure), the amount of organic matter in the soil increases and with it the amount of soil organic N. If the cropping practices are constant over long periods of time (years), the soil organic matter (SOM) level will reach some steady state, that is, SOM no longer increases. Given the fact that many fields have routinely received dairy manure for ten or more years, it is reasonable to assume that long-term forage production fields are in a “quasi” steady-state condition, that is, the annual amount of organic N that is mineralized is approximately equal to the average annual amount of organic N applied or generated as biomass (but not removed at harvest).

While at steady state over the long-term, the soil organic nitrogen pool is potentially very dynamic over the short-term with diurnal (daily), weekly, and seasonal changes in SOM and soil organic N content. Organic matter and organic N dynamics, primarily the mineralization rates at these shorter time-scales control the production of plant-available and leachable forms of nitrogen.

In California, few long-term field studies have been conducted in which soil organic matter and organic N levels have been monitored. One such study currently in progress at UC Davis has shown that high inputs of organic matter (from compost and cover crop residues) in an organic cropping system have increased total N content of the top 12 inches of a loam soil over a 10-year period by 804 lbs N per acre, compared to only 70 lbs N per acre in a four-year conventionally fertilized crop rotation (Poudel et al., 2001). The annual change in organic N storage in the organic cropping system was therefore less than 100 lbs N per acre-yr.

5.7 Nitrogen Rate Guideline: Putting It All Together

5.7.1. Agronomic Rate

The previous sections discussed the various components of the nitrogen cycle in manured field soils assuming good agricultural practices. From an agricultural planning and management perspective this information provides the basis for determining the agronomic rate of N applications (from application of nitrogen fertilizer, manure N, and other N sources). “Agronomic rate” refers to the minimum N input needed to sustain a successful and profitable crop production. In considering land application of organic manure, it is meant to stand in contrast to a “(waste) disposal rate.” The agronomic rate represents the supplements that make up the total N nutritional needs of plants and should take into account N from all sources (including inputs of commercial fertilizers, irrigation water N, fixation of atmospheric N and indigenous soil N) and sinks (including losses of nitrogen through denitrification and ammonia volatilization). The agronomic rate may also include an allowance (additional N) to compensate for field non-uniformity and uncertainty about the crop response.

In commercial production, agronomic rates are not a generalized and fixed N input; instead, they are site- and case-specific quantities depending on production goals, expected N sources and sinks of the soil, and crop N requirements. In the spirit of environmental protection, the off-site and downstream environmental impacts of land application should also be factored into the agronomic rate. The agronomic rate represents the N input for the best biomass production without causing undo groundwater pollution and/or degradation of downstream water quality.

In well-managed fields, application of N according to the agronomic rate would achieve a satisfactory production goal while minimizing the potential for nitrate leaching. In this context, the agronomic rate may be generalized. Broadbent and Carlton (1979) reported that with increasing N fertilizer rate, there was only a slight increase of nitrate in the soil profile available for leaching when fertilizer N input was kept at or below the agronomic rate. When the N fertilizer inputs exceeded the requirement for maximum crop yield, the residual nitrate concentration in the soil profile increased sharply.

If we use the best estimates summarized in the previous sections to account for all the N sinks in the type of crop treatments discussed above, the total N loss in terms of the N applied may be tallied to derive an achievable apparent nitrogen recovery (ANR) and in turn, a nitrogen input requirement (NIR) which is then the agronomic rate for dairy manure applications (Table 5-4). In this assessment, we assume that dairy wastewater from storage lagoons is applied on a low organic matter sandy or loamy sand soil along with irrigation water for corn-winter forage double cropping.

In computing the N losses, we suggest that ammonia losses from plants should not be included. Francis et al. (1993) report those to be on the order of 8% to 14%. However, plants can be both a source and a sink for ammonia. When background atmospheric ammonia levels are elevated above the compensation point, as would often be the case in the vicinity of a dairy, plants are more likely to be a sink rather than a source of ammonia. Furthermore, denitrification has often been estimated by researchers without considering plant losses of ammonia, and in some studies, researchers may unwittingly have included the latter.

When plant ammonia losses are not included, Table 5-4 shows that 28% to 48% of the applied N is lost to sinks other than harvest removal. The apparent nitrogen recovery (ANR), which represents the percentage of applied N taken up by plants for harvest removal, will be the remainder and equals 52% to 72%. The nitrogen input requirement (NIR), which indicates the agronomic rate plus other inputs, is the reciprocal of ANR and is equal to 1.39 to 1.92 (139 to 192%) times the crop N harvest removal.

Detailed dynamic simulations of the nitrogen cycling in California forage systems using ENVIRO-GRO (Appendix E) also showed that crops can successfully be grown with N applications in a range of 1.2 to 1.4 times plant N removal. The simulations are based on physico-chemical computations of root zone water, plant growth, and nutrient cycling that account for the full dynamics of the water and nitrogen cycle in the root zone. The model accounts for irrigation efficiency and non-uniformity, timing of N applications, and potential crop stresses due to water or nutrient deficiencies. The simulations did not include soil or plant NH₃ losses. Adding soil NH₃ losses of 5% of applied N (but not plant NH₃ losses) to the simulation changes the simulated NIR to a range from 1.3 – 1.65, only slightly less than the N input requirement of 1.4 – 1.9 times crop harvest removal estimated from Table 5-4.

Table 5-4: Range of estimated losses of N inputs to forage crops in the Central Valley based on literature values.

Pathway	Estimated N Loss (% of Applied N)		Remarks
	Minimum	Maximum	
Denitrification	8	28	Meisinger and Randall (1991)
Soil NH ₃ Loss ¹	5	10	10 –20% of the applied NH ₄ -N
Leaching ³	15	10	Hanson et al. (1999)
TOTAL	28	48	

¹Assumes 50% of N in the applied dairy wastewater is NH₄-N. ²Based on estimated loss from corn of 40 - 70 lb N/acre (Francis et al., 1993) 20 - 35 lb N/acre for winter forage; and a total of 750 lb N/acre for all external N inputs for the two crops. ³Leaching is minimal in fine-textured soils – the condition leading to denitrification. Therefore, the minimal leaching is paired with maximum denitrification.

5.7.2 Field Examples of NIR and ANR

How do the above estimates of “practically achievable ANR and NIR” compare to field studies and simulations reported in the literature? There are many reports of the crop recovery of a single application of N fertilizer; considerably fewer such studies with a single application of animal manure; and very few studies in which N use efficiency over several years has been calculated. Some investigators have used modeling or regional crop yield and fertilizer usage figures to estimate ANR. Most field studies indicate that NIRs of 140% - 150% are achievable in well-managed agricultural systems (Appendix F), well within the range derived above.

5.7.3 Field Nitrogen Balance

Figure 5-3 shows the result of a hypothetical field N budget, based on realistic transfer coefficients for the major N transformations and loss pathways. In the exercise, the crop harvest N removal, manure characteristics (not shown), and commercial fertilizer rate, along with the

rate coefficients, were arbitrarily chosen but are within the typical ranges in the published literature. It was assumed that the soil organic N content was in a steady state. The budget balances N by calculating the rate of manure N that must be applied to achieve the desired crop N uptake and harvest removal.

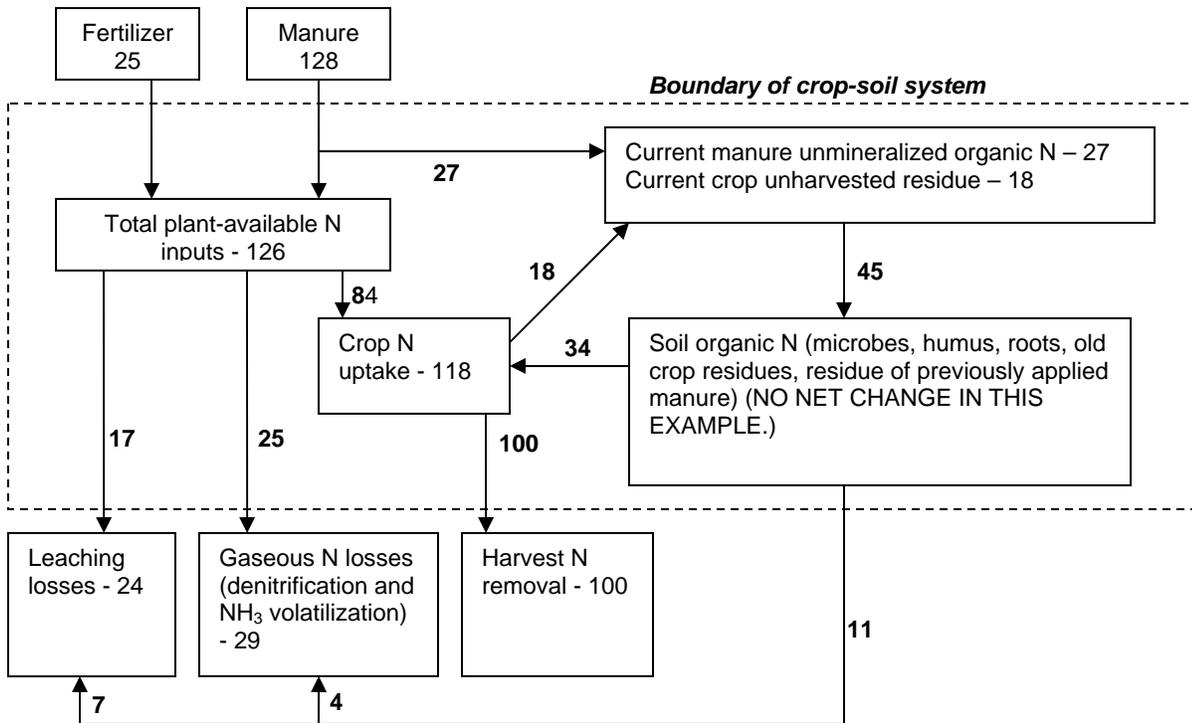


Fig. 5-3: Hypothetical N balance for a forage crop fertilized with dairy manure and commercial fertilizer. Based on 100 lbs N removed in harvested crop. N losses by denitrification, leaching, and manure ammonia volatilization are set at 15%, 15%, and 10%, respectively. N in harvested crop equals 85% of total plant N uptake. Direct loss of ammonia from plants is assumed to be zero.

In this example, the 100 lbs crop harvest N removal is achieved by applying a total of 153 lbs N from external N inputs (i.e., NIR = 153% or 1.53). This is equivalent to an apparent nitrogen recovery (ANR) of 65% ($1/1.53 \times 100$), which is within the range of ANR reported in the scientific literature for various cropping systems. In this scenario, loss of N by leaching is restricted to 15% of N inputs. The results presented here undoubtedly are subject to a great number of uncertainties. It nevertheless demonstrates that the upper end of computer-simulated optimal N loading rates of 1.4 to 1.65 times the crop N harvest removal are practical and, based on field observations, achievable if the production field is properly managed.

5.8 Needs for Further Study

The ANR (Apparent Nitrogen Recovery) and the corresponding NIR (Nitrogen Input Requirement) for dairy manure applications are dependent on the specific agronomic and environmental circumstances at a particular location. While it is not practical to evaluate the ANR and NIR case by case, it may be appropriate that ANR and NIR targets be established for sub-regions, rather than generalizing over the entire Central Valley. Through this approach, the Central Valley may be subdivided into operationally homogeneous regions according to potentials for denitrification, ammonia volatilization, crop N uptake, and nitrate leaching. In Shaffer and Delgado (2002), the merits of this approach were elaborated. Pratt (1979) and Coppock and Meyer (1980) proposed an index for judging the “relative sensitivity” of irrigated croplands to leaching of nitrate. The index included factors relating to soils, crops, irrigation and climate. The UC Center for Water Resources has recently completed development of a regional nitrate leaching hazard index that classifies soil series in irrigated lands in the southwestern US according to the potential for nitrate leaching and denitrification (www.waterresources.ucr.edu).

For improvements, additional and more concise information is needed in the following areas:

- Accurate estimates on the extent of N losses through the various pathways:
 - Denitrification in soil profile in relation to soil texture and irrigation water management
 - Denitrification potential in the deeper unsaturated zone and in groundwater
 - Ammonia volatilization when dairy manure is applied on cropland
 - Nitrate leaching in relation to the irrigation and leaching fractions
 - Potential direct N losses through plants
- Role of alfalfa in the overall N budget of the dairy manure-irrigated forage production system;
- Long-term organic N accumulation and mineralization in dairy manure-receiving fields;
- Fate and transformation of N in solids separated from the liquid manure stream.

5.9 Summary

Both field and modeling studies reviewed and implemented for this report consistently show that the N input requirements for forage crops will generally be in the range of 140% to 165% of the crop N harvest removal, assuming that the manure application would consist of lagoon water which is approximately 75% $\text{NH}_4\text{-N}$. As stated above, inputs include not only manure and fertilizer N but also atmospheric N sources and nitrate present in irrigation water. Investigations of the crop N recovery in several field experiments showed that the appropriate N loading rate that minimizes N leaching and maximizes N harvest is between 140 to 150% of the N harvested. Computer models indicated a somewhat larger range of 140% to 165%. While field studies provided important feedback on loss pathways and loss rates as well as mineralization rates, model simulations were well suited to study the dynamic behavior of the soil nitrogen pool and its interaction with the crop N uptake. Simulations are particularly valuable to understand the role of various loss pathways. Field mineralization, volatilization, and denitrification rates for

specific field conditions can be obtained from detailed field and laboratory studies using standard model calibration and validation approaches.

The combined evidence from laboratory, field, and modeling studies indicates that precise nutrient management, while plausible in principle, may be problematic when implemented in full-scale production systems, as it requires careful timing of the N applications, close monitoring of the amount of N and water inputs, and best management of crop production. More importantly, the growers must show flexibility to make necessary adjustments on N inputs during the course of a growing season to achieve satisfactory results.

With respect to the potential for groundwater degradation, all of the computations and field observations point to a fundamentally critical issue: Given that practically achievable leaching fractions in border check and furrow systems are 15% to 30%, nitrate leaching is at best in the range of 10% to 15% of the N applied. The corresponding nitrogen input requirement is 140 to 165% of N removal at harvest. At annual crop yields that typically remove 400 – 600 lbs N ac⁻¹ yr⁻¹, input requirements will be in the range of 560-990 lbs N ac⁻¹ yr⁻¹. Hence, nitrate-nitrogen leaching losses – under optimal irrigation and nutrient management – will be in the range of 55 to 150 lbs N ac⁻¹ yr⁻¹. Assuming recharge rates in irrigated systems of 1 – 2 acre-feet per acre per year (300 – 600 mm per year), the nitrate concentration in the leachate is in the range of 10 to 55 ppm (mg L⁻¹) NO₃-N, which is at or above the regulatory limit for drinking water quality (10 mg L⁻¹) and at or significantly above the average measured leachate value for other California farming systems (15 mg L⁻¹, Rible et al. 1979). The potential for denitrification in the unsaturated zone below the root zone (not considered in this report) and within the Central Valley aquifers therefore becomes a key factor in determining, whether such (optimal) leaching water quality conditions will still cause groundwater degradation or whether denitrification naturally attenuates nitrate levels to non-degrading levels.

Chapter 6 - Phosphorus and Potassium in Manure Applications

6.1 Introduction

With respect to land application, the relatively fixed composite chemical composition of manure has critical implications for nutrient management that are unlike those in commercial fertilizer systems. The fact that manure is a composite fertilizer, containing nitrogen (N), phosphorus (P), and potassium (K), means that the N:P:K ratios of the fertilizer application is set by the wastewater composition in the storage lagoon (which, in turn, is controlled by feed rations and water handling within the animal systems of the dairy). The control of these nutrient ratios is therefore outside the control of the nutrient manager. Hence, nutrient management that maximizes the benefit of one nutrient, customarily N, may mean that other constituents may be over-or-under-applied. In this respect, the most concern raised by optimizing the N inputs (rather than P or K) has been drawn about potential effects due to overloading of P and K. Losses of the beneficial P and K as plant nutrients may be minor issues. Where surface runoff or tile-drain discharge occurs to nearby surface waters, phosphorus can be of concern, since it leads to the eutrophication of surface water, and some states are setting limits on how much may be applied. Overapplication of K may contribute to the salt load. Hence the question, whether applications of P and K should be limited. If so, what should the limit be and under what circumstances?

6.2 Overview

It has been shown in many locations throughout the United States and elsewhere that when livestock manure is applied at rates needed to meet the crop nitrogen requirement, phosphorus is often applied at higher rates than needed by the crop and may build up in the soil (see for example, Eghball and Power, 1999). In some cases this will lead to transfer of P to surface waters in runoff.

A cow's diet consists of plant material of various forms and mineral supplements. Rarely would a mineral supplement contain nitrogen, but to supplement with materials that contain phosphorus is common. Therefore, the total amount of phosphorus inputs into the diet in relationship to the nitrogen can be controlled to be in the same proportion or higher than that taken off in plant material. If nitrogen and phosphorus excretion were in a similar proportion, then overall, phosphorus would be excessive for plant removal because 1) non-plant sources of phosphorus are added to the diet and 2) nitrogen is subject to losses primarily due to volatilization while phosphorus is not. Potassium is similarly conserved in relationship to nitrogen. Potassium, unlike phosphorus, however, can be taken up in the crops, especially winter cereals, at rates that far exceed crop requirements and these excessively high concentrations often result in undesirable animal health problems.

Few California studies are available that include measurements of the balance of P and K on a whole-farm basis. The few data sets that exist address only lagoon water and do not consider the additional P that may be present in dry manure and in the solids collecting in the bottom of the

lagoon. If these solid materials are applied to land receiving lagoon water, P and K (in addition to N) must be taken into consideration to evaluate the nutrient status.

A data set was compiled as part of a University of California dairy BIFS (Biologically Integrated Farming Systems) study conducted on individual forage fields on nine dairies in the San Joaquin Valley. The data may be used to illustrate the mass balance of P and K in the forage production fields that received the dairy wastewater. (Tables 6-1 and 6-2).

Table 6-1: *The phosphorus (as P₂O₅) balance of BIFS dairies during 2000¹.*

Dairy	Cropping	Conventional ²		Improved ³	
		Input (lbs/A)	Removal (lbs/A)	Input (lbs/A)	Removal (lbs/A)
1	Wheat + Corn	371	332	522	353
2	Corn	366	180	329	246
4	Triticale + Corn	194	332	179	322
6	Corn	266	130	156	176
7	Corn	120	204	120	208
8	Wheat + Corn	524	318	809	324
11	Corn	329	282	329	284

¹Data from Appendix D, "Integrating forage production with dairy manure management in the San Joaquin Valley", Sept. 2001, Annual Report (Year 2) to UC Sustainable Agriculture Research & Education Program, Davis, CA. These data are best estimates based on a combination of actual measurements and estimates or substitutions for missing data. They do not represent a dairy nutrient balance because they do not include the corral manure and pond sludge, which may have a different N:P:K ratio.

²Conventional fertilization practice with N from chemical fertilizer and dairy manure.

³No commercial fertilizer; N from dairy manure only.

Table 6-2: *The potassium (as K₂O) balance of BIFS dairies during 2000¹.*

Dairy	Cropping	Conventional ²		Improved ³	
		Input (lbs/A)	Removal (lbs/A)	Input (lbs/A)	Removal (lbs/A)
1	Wheat + Corn	1303	1287	1836	1287
2	Corn	769	656	750	706
4	Triticale + Corn	656	1030	604	731
6	Corn	844	424	506	487
7	Corn	506	643	506	714
8	Wheat + Corn	1040	1123	2944	1547
11	Corn	n/a	771	1209	701

¹Data from Appendix D, "Integrating forage production with dairy manure management in the San Joaquin Valley", Sept. 2001, Annual Report (Year 2) to UC Sustainable Agriculture Research & Education Program, Davis, CA. These data are best estimates based on a combination of actual measurements and estimates or substitutions for missing data. They do not represent a dairy nutrient balance because they do not include the corral manure and pond sludge, which may have a different N:P:K ratio.

²Conventional fertilization practice with N from chemical fertilizer and dairy manure.

³N fertilizer from dairy manure only.

In most dairies, P and K inputs to the forage production system from dairy manure were comparable to the amounts removed in the crop harvest (Dairies 8 and 10 were notable exceptions). As N inputs of the BIFS dairies were significantly higher than the N removal

through the harvest of forage biomass, there is significant incentive to lower the N inputs by reducing the amount of manure applied. However, lower manure applications would significantly decrease P and K inputs.

In another study, Campbell-Mathews followed nutrient application and removal on the same field near Hilmar, Merced County, for three years. In this study, manure N application was closely matched to crop N uptake (Table 6-3). In this case, the P applied was only slightly less than the P removed, while the potassium applied was generally well above the amount of the potassium removed in the harvest.

Table 6-3: *Nutrient application and removal of nitrogen, phosphorus and potassium by corn silage and cereal forage on a dairy near Hilmar, CA.*

Year	Crop	Applied Nitrogen, lbs/A				Remo ⁵	P ₂ O ₅ , lbs/A		K ₂ O, lbs/A	
		NO ₃ -N ¹	NH ₄ -N ²	Org-N ³	Total	N, lbs/A	Appl ⁴	Remo ⁵	Appl ⁴	Remo ⁵
2001	Corn	41	185	90	316	165	66	49	754	177
2002	Corn	41	200	188	429	232	91	104	683	356
2003	Corn	39	190	125	354	253	55	*	388	*
2004	Corn	24	194	164	382	213	11	66	452	305
2002	Cereal	36	90	47	173	118	44	65	262	322
2003	Cereal	36	117	105	258	187	36	62	393	307
2004	Cereal	8	112	71	191	137	35	53	348	274

¹ Nitrate-N in irrigation water

² Ammonium-N in lagoon water

³ Organic-N in lagoon water

⁴ Appl = applied

⁵ Remo = removed

* Missing data

Neither the BIFS data nor the Hilmar single field data are representative of the large variability in liquid manure composition observed on the respective dairies over the course of a year. Unless solids buildup in a pond is managed to be negligible using agitation or some other method, a significant fraction of the non-soluble phosphorus may also build up in the pond sludge. At some point, this sludge will be land applied. Finally, the above examples only consider liquid manure, but do not take into account contributions from dry manure and separated solids. These materials can be expected to contain significant amounts of phosphorus but smaller amounts of potassium.

In a different study, Campbell-Mathews et al. (2001a) surveyed the nutrient contents of dairy wastewater lagoons in the Central Valley. Assuming crop harvest removal of 105 lbs P₂O₅ per acre (118 kg ha⁻¹) and 250 lbs K₂O per acre (281 kg ha⁻¹) in 30 tons fresh weight per acre (67 mT ha⁻¹) of harvested silage corn, and assuming that these liquid manures had been used to meet N uptake requirements, 21% of the reported cases would have resulted in applications exceeding the expected crop uptake of P₂O₅, and 75% of the cases may have exceed the expected crop uptake of K₂O. This hypothetical exercise assumed that the crop harvest N removal is met with

the total ammonium and 50% of the organic nitrogen in each of approximately 130 liquid manure samples.

In monitoring data from a shallow groundwater monitoring well network (Harter et al., 2002), dissolved total phosphorus in monitoring wells downgradient from fields averaged just above 1 mg L⁻¹ and much less than 1 mg L⁻¹ in monitoring wells downgradient from storage lagoons and corrals. The higher concentrations underneath fields is likely due to the low sorption capacity of the sandy soils at the particular sites and the lack of organic material at the soil surface (relative to the storage lagoon and corral surfaces). Phosphorus tends to strongly sorb to clay particles and organic matter. Hence, even in a sandy aquifer, P is unlikely to be transported large distances off-site. However, under tile-drainage conditions or in the presence of streams in the immediate vicinity of the dairy, phosphorus concentrations at the observed levels may significantly impact surface water quality if groundwater discharges into surface water bodies. It is unknown, whether similar levels of dissolved total P would be observed in regions where soils have higher silt and clay content or in regions with deeper water table.

Soil and tissue sampling could be used effectively to monitor excesses and deficiencies of both P and K. Currently, many dairymen are observing excessively high K concentrations in forages (>3.0%) where lagoon water or manures have been applied. This may lead to depressed dietary magnesium levels, affect animal health and in severe cases cause death of cattle, particularly lactating or pregnant cows (Grunes, 1973).

6.3 Summary

Currently available literature data and the limited amount of recent research in the San Joaquin Valley on the nutrient cycle of P and K in dairies does not allow for a conclusive finding on the question of phosphorus and potassium status in field soils used for application of animal waste. However, trends are certainly apparent that dairymen will need to monitor the increased application of both P and K to determine the impact on forage nutrient levels and animal performance.

Chapter 7 - Salts in Dairy Manure and Salinity Issues in Land Application

7.1 Introduction

In 1973, the UC Committee of Consultants Water Quality Task Force estimated that dairy cows in the Santa Ana Basin, Central Valley and North Coast excreted 1.85, 1.56, and 1.97 lbs (840, 708, and 894 g) of dissolved mineral salts $\text{cow}^{-1} \text{day}^{-1}$, excluding the N containing mineral constituents. In 1973, an average dairy cow was fed 42 to 44 lbs (19 to 20 kg) of dry matter (DM) daily. In 2003, a typical cow was fed 44 to 51 lbs (20 to 23 kg) of DM daily, with some lactating groups with high milk production being as high as 66 lbs day^{-1} (30 kg day^{-1}). While the composition of the rations has changed over this period, the basic ingredients that constitute the rations have remained similar. The inorganic mineral contents of the feed ingredients such as alfalfa hay, grains and others also remained relatively constant. Salt excretion levels are therefore expected to have increased proportional to the DM consumption.

In this chapter, we examine currently known excretion levels based on dietary considerations (section 7.2), and various estimates of field salt loading (section 7.3). Section 7.4 considers the environmental effects of salt loading and section 7.5 the effect of dairies on the regional salt balance in the Central Valley. These last two sections address the question of what constitutes a “reasonable” salt loading rate.

7.2 Salt Intake and Excretion by Lactating Dairy Cows in California

7.2.1 Salt Excretion by Lactating Dairy Cows in California

Salts and minerals in dairy manure derive primarily from minerals in the feeds that constitute the ration, and mineral constituents added to the ration to meet the mineral requirements of the target animal group. While a dairy cow may drink up to 50 gallons (190 l) of water day^{-1} , the dissolved mineral load from the drinking water itself is relatively minor (less than 5%) and not expected to contribute significantly to the overall load, except in exceptional cases. For example, Castillo et al. (2005) reported, based on a survey of lactating dairy cows on 51 randomly selected dairy farms in the California Central Valley, that an average of 16% and 8% of the Na intake and Cl intake, respectively, came from drinking water.

There have been no organized survey studies of dairy rations in California, save the groups of lactating cows described in Chapter 2. Even here, the ability to estimate salt intake is limited to the groups in the Petaluma, Tulare and Modesto areas due to a lack of definition of the added mineral premix in the Chino herd. Currently, salt intake and excretion data are only available for specific ions, namely sodium (Na^+), potassium (K^+) and chloride (Cl^-). Excretion rates for other salts are currently not available although they can be estimated for specific rations (chapter 2).

Based upon the average of the three herds (six groups of cows) the average level of Na^+ , K^+ , and Cl^- in the rations was 2.99% of DM. Assuming an average DM intake of 51 lbs (23 kg) cow^{-1}

day⁻¹, average Na⁺ + K⁺ + Cl⁻ intake would be approximately 1.52 lbs (688 g) cow⁻¹ day⁻¹. According to mass balance calculations on these groups of lactating dairy cows described in Chapter 2, the UC Dairy Animal Waste Estimator calculates that 82, 91, and 76% of Na⁺, K⁺, and Cl⁻, respectively, will be excreted in feces and urine or, on average, about 85% of the total Na⁺ + K⁺ + Cl⁻ intake. Thus of the 1.52 lb (688 g) d⁻¹ consumed, about 1.29 lbs day⁻¹ (585 g day⁻¹) [or 392 lbs (178 kg) cow⁻¹ per 305 days lactation] of Na⁺ + K⁺ + Cl⁻ will be excreted. That these values are less than values reported by the 1973 UC Committee of Consultants Water Quality Task Force partially reflects the fact that the 1.29 lbs day⁻¹ is Na⁺ + K⁺ + Cl⁻ only and not total salts. Based on dietary considerations, total salt excretions are typically more than 50% higher and as much as 100% higher than the Na⁺ + K⁺ + Cl⁻ excretions alone (see, for example, the survey reported by Castillo et al., 2005).

7.2.2 Salt Intake and Excretion by Dry Dairy Cows in California

There are no equivalent California data for dry cows comparable to those presented for lactating cows in Chapter 2 and above. However, in general, dietary salt levels are lower (by approximately 20%) in rations of dry dairy cows and feed intake levels are substantially less. However there is no milk being exported, hence there is no salt export in milk. Thus, for average California dry cows consuming 26 lbs (12 kg) of DM cow⁻¹ day⁻¹, a reasonable average Na⁺ + K⁺ + Cl⁻ intake would be 0.68 lbs (287 g) day⁻¹ of which 95% would appear in manure, or 0.60 lbs (273 g) day⁻¹ [35 lbs (16 kg) cow⁻¹ day⁻¹ dry period].

7.2.3 Factors that Affect Salt Intake by Dairy Cows in California

Salt intake by dairy cows can be expected to vary to a great extent among cows, and among groups of cows on different dairies, due to differences in DM intake as well as dietary salt levels. Differences in dietary salt levels can be caused by differences in the salt levels (especially K) of the forages in the ration as well as formulated minimum levels of salts among consulting nutritionists. In addition, it is still not uncommon to find 'free-choice' salt (NaCl only or trace mineral enhanced NaCl) provided either loose or in pressed blocks. This practice can dramatically increase salt intake and excretion.

7.3 Field Salt Loading

Salt load onto the field is defined as mass of total mineral salts (total dissolved solids, TDS) added to a unit of land area over a given time period. In dairy wastewater, the TDS is composed predominantly of the major inorganic cations (Ca²⁺, Mg²⁺, Na⁺, K⁺, NH₄⁺) and anions (Cl⁻, SO₄²⁻, HCO₃⁻, and NO₃⁻) but also includes a significant amount of organic anions, most of which will ultimately mineralize to the bicarbonate anion (HCO₃⁻). In irrigation water, more than 95% of the TDS is due to Ca²⁺, Mg²⁺, Na⁺, K⁺, Cl⁻, SO₄²⁻, HCO₃⁻.

Salt loads are unavoidable on irrigated land as the irrigation water invariably contains dissolved minerals which will be concentrated when the applied water evaporates and transpires. Salt inputs (TDS) from the irrigation water for year-round production of forage crops in the Central Valley is on the order of 2858 lbs ac⁻¹ (3,200 kg ha⁻¹). This estimate is based on the assumption

that approximately 1 m (3.3 ac-ft ac⁻¹) of irrigation water is applied per year and that the electrical conductivity (EC) of irrigation water is on the order of 0.5 mmho cm⁻¹ (dS m⁻¹). This EC corresponds to a total dissolved solids concentration of approximately 320 ppm (mg L⁻¹). In most irrigation water in the eastern San Joaquin Valley and in the Sacramento Valley, the majority of dissolved solids are Ca²⁺, Mg²⁺, HCO₃⁻, and SO₄²⁻. Irrigation water, particularly surface water, is relatively low in Na⁺, Cl⁻.

On dairies, field salt loading and subsequent salt loading to groundwater (the amount of total dissolved solids annually recharged to the water table per unit land area) can be estimated by various methods. We estimated approximate field salt loading on manure-treated land a) based on average excretion rates of Na⁺ + K⁺ + Cl⁻ and nitrogen and nitrogen application rates in fields (holistic, farm-based mass-balance approach) and b) based on field reconnaissance data of liquid manure water quality (lagoon water quality approach). Using chemical speciation models, we derived groundwater loading rates from the land application and crop nutrient uptake data and compared those to measured groundwater quality data.

7.3.1 Mass Balance Estimation

The use of dairy manure in nutrient management imposes a heavier salt load on croplands receiving the manure than on croplands receiving commercial fertilizer only. Under the year-round forage production scenario for the Central Valley, the crop nitrogen uptake may be much as 268 – 625 lbs N ac⁻¹ yr⁻¹ (300 - 700 kg N ha⁻¹ yr⁻¹). Table 7-1 gives the amount of Na⁺, K⁺, and Cl⁻ salts excreted and applied if the nitrogen applied is from dairy manure only. The values in Table 7-1 assume that manure application rates match the recommended average NIR of 1.4 to 1.65 (see chapter 5) and that the total N losses in flush lanes, corrals, and the lagoon are in the range of 22% to 50% (somewhat broader than the range given in chapter 4). Low values assume lowest estimated N losses and lowest average NIR, while high values assume highest estimated N losses and highest average NIR. Salt excretion is based on the estimates given above.

Table 7-1: Approximate range of nitrogen and Na⁺ + K⁺ + Cl⁻ excretion [kg/ha/year] as a function of the annual crop nitrogen uptake [kg/ha/year]. Ranges of NIR and N losses assumed are described in chapters 4 and 5.

	Annual Crop N Uptake, lbs ac ⁻¹ yr ⁻¹ (kg ha ⁻¹ yr ⁻¹)				
	268 (300)	357 (400)	447 (500)	536 (600)	625 (700)
N excretion, lb/A/yr (kg/ha/year)	482-884 540-990	643-1179 720-1320	804-1473 900-1650	964-1768 1080-1980	1125-2063 1260-2310
Salt excretion, lb/A/yr (kg/ha/year)	616-1125 690-1260	813-1500 910-1680	1018-1875 1140-2100	1223-2250 1370-2520	1429-2625 1600-2940
Lactating plus Dry Cows equivalent head/A (head/ha)	1.4 – 2.6 3.5 – 6.5	1.9 – 3.5 4.7 - 8.6	2.4 – 4.4 5.9-10.8	2.8 – 5.2 7.0 - 12.9	3.3 – 6.1 8.2 – 15.1

Unless manure solids are exported from a dairy facility, the long-term salt application rates within the dairy are equal to the salt excretion rate. Other than through solids export, there are no net losses of Na^+ , K^+ , and Cl^- (or other salts) in the dairy manure pathway. On facilities with significant solids separation, solids may be temporarily recycled for bedding, but will ultimately be applied to fields (see section 3.3.3 of this report). Occasionally, manure is applied to non-forage crops (e.g., cotton), but the salt export through the sale of these crops is relatively small.

Given the values in Table 7-1, the salt load in high production forage systems receiving dairy manure application is expected to be approximately 50% to over 100% higher than on land in regular crop production (with irrigation water as the sole sources of salinity). Note that Table 7-1 does not include all salts. Ca^{2+} , Mg^{2+} , SO_4^{2-} , HCO_3^- and nitrogen salts in manure also contribute significantly to the salt loading. On the other hand, crops have a significant capacity to take up potassium and nitrogen salts (in form of NH_4^+ or NO_3^-) (see chapter 5).

7.3.2 Groundwater Loading Estimation using Chemical Speciation Model

The above estimates represent salt loading to the land surface. The actual salt inputs to groundwater will be dependent on the compositions of the inorganic chemical constituents in the manure, the capacity of the crop to take up salts (predominantly N and K), and chemical precipitation-dissolution reactions in the soil. Under appropriate conditions, certain minerals may precipitate out of the solution and thus reduce the inputs. It is also possible that the oxidation of ammonia to nitrate will induce dissolution of precipitated minerals, thus increasing the amount of salts being leached.

The calcium and magnesium ions commonly found in the lagoon dairy wastewater are especially susceptible to precipitation reactions. The amount of net precipitation/dissolution and the chemical composition of the water leaching out of the root zone was estimated based on known chemical thermodynamics. To assess the geochemistry of soil water in manure applications, we used the chemical speciation computer program WATSUIT for Windows (<http://envisci.ucr.edu/faculty/laowu/default.htm>). The model estimates typical precipitation/dissolution rates and the chemical composition of root zone leachate given a typical composition of diluted manure water applied to a summer and also to a winter crop at the rates recommended in chapter 5. WATSUIT is a steady state model and considers only soil water constituents, but not soil chemical properties. Given those limitations, the program provides a good first approximation. In practice, transient conditions may play an important role and more complex, coupled models that simultaneously consider pore water flow dynamics, solute transport dynamics, and geochemical reactions may be needed to fully evaluate the dynamics of these geochemical processes.

Table 7-2 summarizes several generic examples of annual salt loading to groundwater for four typical irrigation scenarios in the San Joaquin Valley, adjusted for plant K uptake. These estimates were obtained from steady-state WATSUIT computations (see Appendix J for details). The results suggest that the use of dairy wastewater, under proper nutrient management practices and with manure as the only source of fertilizer, increases the annual salt loading by 2500 to 3500 kg ha^{-1} . The recharge water also is significantly higher in hardness and alkalinity than

recharge water in commercially fertilized crops (without manure). Lime precipitation does not appear to play a significant role in reducing the salt loads to groundwater.

Table 7-2: Salt loading of dairy wastewater application fields after adjusting for K uptake.

Irrigation Water Source	Salt Input, lbs ac ⁻¹ (kg ha ⁻¹)		Annual Salt Loading (kg ha ⁻¹)
	Winter Forage	Summer Corn	
East Side Sources	77 (86)	277 (310)	361 (404)
Wastewater + East Side	1211 (1356)	2040 (2284)	3228 (3615)
West Side Sources	739 (828)	2664 (2983)	3388 (3794)
Wastewater + West Side	1786 (2000)	4279 (4792)	5762 (6452)

[†]Annual Summer Corn/Winter Forage Double Cropping with 250 and 150 lbs per acre of N inputs, respectively; annual water inputs are rainfall 12 inches ((30.48 cm), winter irrigation 10 inches (25.4 cm), and summer irrigation 36 inches (91.44 cm); and leaching fraction is 0.3.

7.3.3 Groundwater Loading Estimation using Groundwater Monitoring Data

Harter et al. (2002) and Harter and Talozzi (2004) measured groundwater salinity in shallow groundwater originating from recharge in corrals, manure storage lagoons, and fields receiving liquid manure. Soils on the tested dairies were sandy with low ion exchange capacity. Their results confirm that the average salinity downgradient from fields receiving liquid manure is approximately twice as high (1.6 mmho cm⁻¹) as in groundwater upgradient from the investigated dairy facilities (0.8 mmho cm⁻¹). Even higher salinity losses were found to occur underneath corrals and lagoons (2.3 mmho cm⁻¹). However, a direct comparison of these study results to the above modeling estimates would not be appropriate. In the groundwater study, “upgradient” groundwater at the measured locations is recharged from commercially fertilized fields and orchards with unknown and variable crop production (and fertilization) rates. Also, few of the manured fields monitored for those studies had been subject to the type of sustainable nutrient management activities suggested in chapter 5. Actual loading rates on individual facilities and in individual fields will vary significantly depending on feed rations, manure and nutrient management, and irrigation practices. To some degree, they will also be a function of soil properties, particularly in soils with naturally higher salinity.

7.3.4 Estimating Groundwater Salt Loading from EC Measurements

The above methods provide estimates and case studies of average or actual salt loading to the field and to groundwater under conditions typical for dairies in the Central Valley of California. For site-specific measurements of groundwater salt loading, it has been common practice in irrigation agriculture to measure the electrical conductivity (EC) of the irrigation water (ASCE, 1990). However, this simple method cannot be applied to estimate salt loading from dairy wastewater applications because the method does not account for nutrient uptake and has not been tested on liquid manures. There are currently no well-established methods (and little research is available) on how to measure or estimate the site-specific groundwater salt loading from applications of dairy manure. Based on our geochemical modeling results and other work reported above, we recommend that future methods for computing annual salt loading from individual fields account for the annual plant NH₄⁺ + K⁺ uptake, which constitute a significant fraction of the manure salts. Mineralization of salts from organic matter also must be considered.

7.4 Agronomic Impacts of Salinity

7.4.1 Overview

In addition to its cumulative effect on groundwater quality, salinity of water may impact its utility for irrigation through impacts on soil properties and on plant physiology. The major cations in typical irrigation waters are Ca^{2+} , Mg^{2+} and Na^+ and the major anions are Cl^- and SO_4^{2-} . Typically, the plants absorb only a small portion of the total dissolved mineral input from irrigation water, so the soil solution concentration of dissolved minerals increases as water is extracted from the root zone and transpired. If the salinity of the soil solution becomes too high, water uptake is reduced and plant growth is impaired as a consequence of the reduced osmotic potential. The negative impact of soil salinity on plant physiology is remedied by applying extra water to leach excessive salts from the root zone. The amount of leaching required depends on the crop tolerance to salinity and the salinity level in the irrigation water. Models are available to simulate the consequences of irrigation water salinity and irrigation water availability on plant growth and on the concentration of salts in the water moving below the root zone. In a long-term sustainable crop production system that does not use manure, it is commonly assumed that the total amount of salt leached below the root zone equals the total amount of salt applied in the irrigation water.

7.4.2 Crop Impacts

Dairy lagoon waters have high concentrations of NH_4^+ and K^+ . These carry electrical current as well as contribute to negative osmotic potential. However, unlike Ca^{2+} , Mg^{2+} , and Na^+ that are only sparingly taken up by the crop, NH_4^+ and K^+ are taken up in larger quantities over the course of the growing season. The uptake of these nutrients lowers the EC and reduces the effects of the negative osmotic potential. Indeed, if the application rate of NH_4^+ and K^+ in lagoon water does not exceed the crop removal rate, they have essentially no salinity impact on crop growth or irrigation management by the end of the growing season.

The question remains as to the osmotic impact of NH_4^+ and K^+ on plant response shortly after application and before much uptake by plants. The following reasoning is used to answer this question. The salinity of irrigation water without plant nutrients is typically used in experiments designed to establish the relationship between salinity and plant response. Nevertheless, plant nutrients are applied to the soil in these experiments to promote plant growth. In these experiments, the plant nutrients are often added in smaller increments relative to expected plant uptake so that the additional “salt effect” is minimized. Applications of fertilizer in the field cautiously consider placing fertilizer close to the seed for greatest early uptake but with low potential for “salt or high ammonia concentration” damage to the seedling. The “salt index” (WFH, 2002) is used along with the likelihood of ammonia toxicity from anhydrous ammonia (Colliver and Welch, 1970), di-ammonium or mono-ammonium phosphate (Alfred and Ohlrogge, 1964) or urea and other fertilizer salts (Cummins and Parks, 1961). Since the amount of starter fertilizer placed with or close to the seed is usually less than 200 lbs ac^{-1} (224 kg ha^{-1}) and “safer fertilizers” are used, the possible deleterious salt effects are likely to be small. Higher rates of fertilizer are placed at deeper soil depths and farther away from the plant to avoid possible salt damage during early growth stages. The small additional osmotic contribution of

the low rates of fertilizer placed at a safe distance from the seedling is not considered to significantly contribute in the experimental salinity evaluation. Because the NH_4^+ and K^+ ions have both been demonstrated to be toxic to plant seedlings, they can not be overlooked as part of the “salinity” of lagoon water and other manure applications. Yet, after these nutrients are taken up by the plant they no longer influence the salinity of the soil.

Analyses of dairy lagoon water sampled by Campbell-Mathews et al. (2001a) reveal that, on average, approximately 58% of the cation equivalents (measured in meq L^{-1}) were NH_4^+ and K^+ . Since both of these cations have been demonstrated to be among the more toxic to plants, particularly in the seedling and early growth stages, further research must be conducted to elucidate the relative salinity-specific ion effects. Also, a more complete evaluation of all the cations and anions making up the “salinity” in soil water following lagoon water applications is necessary. Certainly the more toxic chloride ion is often present in significant concentrations in lagoon water and other manure wastes but the other anion constituents may be considerably less toxic. Application of the undiluted lagoon water has been demonstrated to be injurious to many crops. For agronomic reasons, applications of undiluted lagoon water have therefore occurred either during fallow periods and incorporated into the soil after the soils have dried or by mixing and dilution with irrigation water. If lagoon water is diluted with irrigation water to match the N needs for crop nitrogen uptake, crop injuries are not likely to occur.

7.4.3 Salinity Impacts on Soil Physical Properties

Saline irrigation water and salt-affected soils may cause two undesirable effects on the receiving soils and growing plants. The total electrolyte concentration of the soil solution affects plant growth and the chemical composition potentially affects soil physical properties. Sodium and other monovalent cations cause dispersion of soil colloids; calcium, magnesium and other divalent cations tend to promote flocculation of soil colloids and stability of soil aggregates. Therefore, the proportion of sodium (primary monovalent cations) vs. calcium and magnesium (primary divalent cations) in the soil solution will determine the physical state of soil particles. Soil dispersion contributes to low water infiltration, soil compaction, and reduced aeration. The sodium adsorption ratio (SAR) of water is used as an index of the hazard of irrigation on physical properties of the receiving soils. Mathematically, SAR is calculated as follows:

$$\text{SAR} = [\text{Na}] / \{([\text{Ca}] + [\text{Mg}]) / 2\}^{1/2} \quad [\text{Eq. 7-1}]$$

where $[\text{Na}]$, $[\text{Ca}]$, and $[\text{Mg}]$ are equivalent concentrations of Na^+ , Ca^{2+} and Mg^{2+} expressed in meq l^{-1} .

Although several factors can affect the relationship between SAR and soil physical properties, $\text{SAR} = 15$ is commonly considered as a reference point. Water with SAR values greater than 15 are considered hazardous and unsuitable for irrigation; water with SAR values less than 15 would be incrementally less hazardous in irrigation. Calculations on six waters analyzed by Campbell-Mathews ranging in EC from 2.75 to 10.1 mmho cm^{-1} (dS m^{-1}) resulted in SAR values ranging from 2.0 to 8.5 and the SAR tended to increase with an increase in EC. Use of these waters for irrigation would not be considered detrimental to soil physical conditions.

Establishing the hazard of lagoon waters on soil physical properties is non-trivial and does not follow established SAR guidelines. The NH_4^+ and K^+ are monovalent cations prevalent in the dairy lagoon water. They will have less severe but notable adverse impacts on physical properties of the receiving soils. The usefulness of SAR values in the context of applying liquid dairy manure is therefore unknown. It may be argued that the impacts are transient in nature, because NH_4^+ and K^+ are ultimately absorbed by plants and the adverse effect would disappear except for the acidification that occurs during the conversion of NH_4^+ to NO_3^- (Robbins et al., 1983, 1993, 1996; Robbins, 1984) However, clay dispersion and soil plugging are largely irreversible and can only be corrected by tillage to reform soil aggregates. The impacts of monovalent-dominated water on soil physical properties is not trivial, however transient, and further research is needed to address this issue with more confidence.

7.5 Production of Feed in the Central Valley: Regional Salt Balance Analysis

From a regional salt cycling perspective, the production facilities of a dairy do not generate salts, they are merely part of a regional and national salt cycle. Salts are brought onto the dairy in form of imported feed stuff and exported via milk sales, manure exports, surface runoff, and groundwater leaching. What is the impact of dairies on the regional salt balance?

Feed rations for dairy animals consist of forage (legume and grass hays), plant by-products (almond hulls, beet pulp, citrus pulp, tomato pomace, etc.), grains and whole seeds (barley, corn, oats, wheat, cotton seeds, canola seeds, etc.), protein meals (canola meal, cottonseed meal, soybean meal, fish meal, blood meal, etc.), commercial supplements, mineral and vitamin ingredients, and miscellaneous ingredients. Forage, plant by-products, grains and whole seeds usually constitute 85% to 90% of the dry matter in the rations and contribute to the bulk of the dissolved minerals present in the excreted manure. For example, Ca^{2+} , Mg^{2+} , Na^+ , K^+ , Cl^- , and SO_4^{2-} typically are 0.85% to 1.45%, 0.27% to 0.42%, 0.15%, 1.8% to 3%, 0.34%, and 0.9% of the dry matter in alfalfa hays, respectively. More than 90% of the non-nitrogen related minerals in dairy feeds are excreted.

The eight dairying counties in the San Joaquin Valley (i.e. Fresno, Kern, Kings, Kings, Merced, San Joaquin, Stanislaus, and Tulare) are major production areas of alfalfa hay in the state (Table 7.12). According to the summary of the County Agricultural Commissioners' Crop Reports (California Agricultural Statistics Service, 2003), the gross value of 2001 – 02 year alfalfa hay production in the San Joaquin Valley amounted to \$534 million. For the same period, the gross value of alfalfa hay produced by Southern California counties (i.e. Imperial, Los Angeles, Riverside, and San Bernardino) was \$205 million. Alfalfa hay produced in Southern California is mostly for consumption in the Chino-Corona and Hemet-San Jacinto dairy areas, where approximately 200,000 dairy cows are located. The remainder is exported to dairies in Arizona.

Generally, feeds for dairy cows are produced on nearby farmlands. For dairies in the Central Valley, some of the feed, especially alfalfa hay, may be produced and imported from outside the Central Valley. However, there are few data available on the extent of the importation. Based on the harvested acreage of hay (including green chops) and grains (corn, oat, and wheat) as reported in the 2002 County Agricultural Commissioners' Data, it is reasonable to assume that

the majority of the feed ingredients used by Central Valley dairies are raised on farmlands in the Central Valley (California Agricultural Statistical Service, 2003).

Table 7-3: *Gross Values of Alfalfa Hay Production in San Joaquin Valley and Southern California Counties in 2001 – 02.*

San Joaquin Valley County	Gross Value (\$)	Southern California County	Gross Value (\$)
Fresno	72,220,000	Imperial	137,074,000
Kern	112,180,000	Los Angeles	5,897,000
Kings	50,186,000	Riverside	50,758,000
Madera	32,650,000	San Bernardino	11,214,000
Merced	73,835,000		
San Joaquin	63,430,000		
Stanislaus	38,372,000		
Tulare	90,868,000		
TOTAL	533,741,000	TOTAL	204,943,000

[†]Derived from data in California Agricultural Statistics Service, 2003

In addition to alfalfa, San Joaquin Valley counties have significant acreages of other crops that are major ingredients in dairy animal feed, such as corn silage, corn, winter forage, wheat, almond, tomato, citrus, etc. The production acreages in Fresno, Kern, and Merced counties are used as examples (Tables 7.13 to 7.15). Patterns in other counties are similar. They clearly illustrate the extent of local feed production for dairy cows in the San Joaquin Valley.

Alfalfa hay is fed not only to dairy cows but also to horses and other ruminant animals. According to Dr. Daniel H. Putnam (CE Agronomy Specialist, UC Davis), California has a deficit in alfalfa production and imports probably 3 to 10% of its needs. While anecdotal evidence exists, there are no data to indicate the extent of the forage importation into the San Joaquin Valley. Small feed amounts may be imported to cover seasonal and temporary shortages in supply. We do not expect the imported amounts, especially alfalfa hay, to be significant as the shipping costs and distances will be considerable compared to local production. Imperial Valley and Palos Verde Valley are the closest major forage production areas outside of San Joaquin Valley. One-way shipping distance from these production areas will be 500 miles or longer.

Despite significant local productions, some grains (especially corn) and protein meals (such as soybean and canola meal) for dairy feeds may be imported from the Mid-West and Canada. These two ingredients typically constitute 10 to 15% of the dry matter in the feed while forage typically is less than 50% of the dry matter in the feeds. The mineral contents of grains and protein meals are considerably lower than those of the forages. For example, the whole corn grain contains 0.03%, 0.03%, 0.37%, 0.05%, and 0.12% of Ca²⁺, Na⁺, K⁺, Cl⁻, and SO₄²⁻-S, respectively. As a result, the salt contribution of the grains and protein meals in the dairy feed will be far less significant than the forages and agricultural by-products which are locally produced.

Table 7-4: Production Acreages of Selected Crops in Fresno County in 2002.

Crop	Harvested Acreage (Acre)
Alfalfa hay	76,300
Hay (others)	26,100
Barley	8,600
Corn (grain)	1,790
Corn (silage)	24,000
Wheat	61,000
Tomato (processing)	115,000
Cotton ¹	232,640
Almond ²	63,450

Derived from data in 2002 Fresno County Crop Report (<http://www.co.fresno.ca.us/4010/crop02/index.html>)

¹Cottonseed 294,490 tons

²Almond hulls 168,000 tons

Table 7-5: Production Acreages of Selected Crops in Kern County in 2002.

Crop	Harvested Acreage (Acre)
Alfalfa hay	148,000
Hay (grain)	20,000
Hay (others)	15,000
Silage/forage	58,500
Barley	9,100
Wheat	74,000
Cotton ¹	130,082
Almond ²	94,067
Sugar beets	3,251

Derived from data in 2002 Kern County Crop Report (http://www.co.kern.ca.us/kernag/crop00_09/crop02/contnts.htm)

¹Cottonseed 133,000 tons

²Almond hulls 230,000 tons

At the regional scale, it therefore appears that when dairy manure is applied on croplands, the overall salt balance of the Central Valley is unaffected. Harvested plants absorb dissolved minerals from the soil, are transported to the dairy, fed to cows, (partially) excreted, then redistributed through land application of dairy manure.

Despite the apparent regional balance, two additional issues must be considered, both of which highlight the localized increase in salt load when comparing animal farming systems (including dairies) to other farming systems:

(1) The production of forage crops for consumption by Central Valley animals replaces the production of food and fiber crops for human consumption. Most foods and fibers are exported from the Central Valley (salt export), while forage crops remain in the Central Valley. Hence, for each animal unit in the Central Valley, there is a net increase in salt that remains in the Central Valley (via excretion and land application).

(2) The concentration of animals in dairies and other animal farming operations means that the salt loading to land is concentrated in the vicinity of dairies, where the manure is most likely to be land- applied, whereas the forage crop production (from where the salts originate) occurs over a much larger land area and is more dispersed.

As a result, salt loading in dairy areas are expected to increase (relative to non-dairy areas) and, over the long run, the salinity of groundwater underneath these areas may be affected.

Table 7-6: Production Acreages of Selected Crops in Merced County in 2002.

Crop	Harvested Acreage (Acre)
Alfalfa hay	78,189
Hay (grain)	31,245
Hay (Sudan)	3,635
Barley	2,660
Wheat	14,417
Oats ((grain)	4,045
Corn (grain)	3,658
Corn (silage)	78,175
Silage (others)	70,194
Sugar beets	4,386
Cotton ¹	54,730
Almond ²	83,535

Derived from 2002 Merced County Crop Report (<http://www.co.merced.ca.us/ag/croprpt02/index.html>)

¹Cottonseed 57,467 tons

²Almond hulls 149,669 tons

7.6 Summary

The excretion of salts can be expected to vary dramatically due to on-farm management decisions and practices. Regardless, daily consumption and excretion of salts will be dramatically lower for dry vs. lactating stock, although even this can be affected by management decisions.

From a dietary point of view, exact salt excretion data are currently unavailable except for Na⁺, K⁺, Cl⁻, and total N excretion. Excluding uncontrolled provision of salt to lactating dairy cows, a summary of six groups of cows on three commercial California dairies suggests that the average lactating dairy cow will excrete 1.29 lb (585 g) day⁻¹ of Na⁺-K⁺-Cl⁻ salts and the average dry cow will excrete 0.63 lb (287 g) day⁻¹ (also Na⁺-K⁺-Cl⁻ salts only). Assuming an annual division of 305 days lactating and 60 days dry, the average dairy cow will excrete 427 lb (194 kg) year⁻¹ of Na⁺-K⁺-Cl⁻ salts. That these values are less than values reported by the 1973 UC Committee of Consultants Water Quality Task Force reflects the fact that comparable data for other salts (Ca²⁺, Mg²⁺, HCO₃⁻, SO₄²⁻) are not available.

Analysis of manure data, geochemical modeling, and observations of groundwater recharge quality in the San Joaquin Valley dairies suggests that the salinity contribution (defined as the mass of total dissolved solids) from manure, under proper nutrient management practices that seek to maximize the use of lagoon water as a source of fertilizer, is on the order of 1786 – 3572 lbs ac⁻¹ yr⁻¹ (2000 – 4000 kg ha⁻¹ yr⁻¹). For comparison, the salt loading from irrigation water alone, depending on the source of the irrigation water, is on the order of 357 lbs ac⁻¹ yr⁻¹ (400 kg ha⁻¹ yr⁻¹) for lower salinity water sources (e.g., Sierra Nevada watersheds) to nearly 4,000 kg ha⁻¹ yr⁻¹ for higher salinity water sources (e.g., State Water Project).

At the regional scale, dairies are only one of several sources of salinity to the Central Valley's groundwater and surface water supply. Locally, they add significant additional salinity to groundwater. The long-term impacts from dairies as well as those from other salinity sources (municipal wastewater treatment plants, food-waste dischargers, etc.) are still not well understood. Increasing salinity in California's waters is an issue that must be dealt with as part of an integrated long-range water resources management plan.

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APPENDIX A: An Example Dairy Manure Output Calculation

Andrew Chang, Peter Robinson

The manure nutrient output of any dairy herd can be evaluated based upon knowledge of the ration fed and based on animal production records. Using the 'UC Dairy Animal Waste Model', developed as a part of this project, estimated manure nutrients generated by an example dairy in the Central Valley of California were calculated. Input data and other information used in this computation can be extracted from production records of well managed commercial dairy operations. This example dairy is not a single operation, but rather a composite of several dairies, and is presented to demonstrate what can be done, rather than to create average manure nutrient output values.

A 1,750 animal dairy (about the California average) was divided into milking, dry and replacement animals, as this is the way that dairy animals are almost always considered on commercial operations. Within each of these categories, animals were further sub-divided into several homogeneous performance groups that possess similar physiological attributes and would be fed the same (or very similar) rations in the same physical groups (i.e., pens) on a commercial dairy (Table A-1). In this example, mature milking cows and dry stock body weights vary between 1,323 to 1,543 lbs (600 to 700 kg). The feed intake of lactating cows varies between 46 to 55 lbs/head/ day (21 to 25 kg/head/day) depending on their milk production and stage of lactation. Dry stock are fed 50% to 60% of what is fed to the lactating cows. For replacement heifers, the body weights are proportional to their ages, from 132 to 1190 lbs (50 to 540 kg) and the feed intakes increase incrementally with body weight.

Based on the specific rations fed (i.e., their ingredients and analyzed or estimated nutrient composition), the rations fed are entered into the program, by animal group as defined in Table A-1, and calculations are made to allocate feed components to body gain or loss and milk production. An example for lactating cows is shown in Table A-2. The daily per animal production of nutrients in manure and urine is then calculated. Summation of input and output of nutrients, by defined animal group in Table A-1, is arithmetically combined to create a whole farm balance (Table A-3) for all nutrients defined in the rations fed to the cattle.

In a well managed dairy, manure and N outputs of milking cows are generally proportional to their feed intake and the manure/feed ratios remain fairly constant regardless of the productive performance of the cows and replacement heifers. However these ratios are impacted dramatically by the level of nutrients in the diet. As nutrient levels increase, manure/feed ratios increase. This site specific approach captures that variation thereby allowing efficiency of nutrient use to be estimated among commercial dairy operations.

Table A-1: Characteristics of dairy animals on an example central valley dairy.

	Milking Cows ¹					Dry Cows ²		Replacement Heifers ³				
	1	2	3	4	5	1	2	Wet	Open 1	Open 2	Bred 1	Bred 2
Number of animals	43	129	268	196	166	83	42	102	205	205	200	107
Milk yield (kg/d)	30.49	34.48	47.64	38.11	27.70	-	-	-	-	-	-	-
Milk fat (%)	4.60	3.60	3.45	3.50	3.60	-	-	-	-	-	-	-
Milk crude protein (%)	3.42	2.95	3.05	3.20	3.45	-	-	-	-	-	-	-
Body weight (kg)	635	590	635	658	681	635	658	60	227	340	453	540
Body condition ⁴ (unit)	3.50	3.50	3.00	3.40	3.70	3.70	3.70	3.00	3.20	3.40	3.70	3.70
Lactation number	2.3	1.1	3.3	2.4	2.2	2.4	2	1	1	1	1	1
Average days in milk	77	110	106	229	345	-	-	-	-	-	-	-
Days pregnant	47	78	82	114	256	250	270	-	-	-	100	220
Calf birth weight (kg)	43	43	43	43	43	43	43	-	-	-	39	39
Dry matter intake (kg/d)	21.98	22.17	24.74	23.91	21.00	12.95	13.55	1.80	4.89	8.58	10.00	10.74

¹ Milking cows are divided into groups by a number to represent the feeding strategies that differentiate them.

² Dry cows are divided into groups by a number to represent the feeding strategies that differentiate them.

³ Replacement heifers are divided into groups ('wet' – milk fed; 'open' – not pregnant; 'bred' – pregnant) by a number within group to represent the feeding strategies that differentiate them.

⁴ Body condition is a visual score that indicates the degree of body fatness where 1 is severely emaciated and 5 is severely obese.

Table A-2: *Daily manure production of milking cows.*

Estimates	Group 1		Group 2		Group 3		Group 4		Group 5	
	Solids	N	Solids	N	Solids	N	Solids	N	Solids	N
Feed Consumed (g/d)	20081	710	20309	695.4	22707	730.1	21929	701.9	19303	599.1
Milk (g/d)	3948	166	3903	159.4	3903	227.7	5369	191.1	4370	149.8
Body retention (g/d)	28	2	80	7	26	2	53	5	191	17
Manure (g/d)	7322	542	7334	528.9	8224	500.0	8098	506.1	7727	432.5
Manure/Intake ratio	0.37	0.76	0.36	0.76	0.36	0.69	0.37	0.72	0.40	0.72

Table A-3: *Mass balance of dairy inputs and manure outputs for a 1,750 animal dairy.*

Category	Macro Element (kg/d)								Trace Element (kg/d)			
	N	P	K	Ca	Mg	S	Na	Cl	Mn	Zn	Cu	Se
Feed input	762.7	101.0	491.6	217.2	93.9	75.8	68.3	113.5	0.89	1.70	0.49	9.21
Exported in milk	175.5	29.0	44.9	38.1	3.3	9.1	12.7	28.1	0.02	0.15	0.01	1.16
Excreted in manure	587.2	72	446.7	179.1	90.6	66.7	55.6	85.4	0.87	1.54	0.48	8.05
Manure/input ratio	0.77	0.71	0.91	0.82	0.96	0.88	0.81	0.75	0.98	0.91	0.98	0.87

APPENDIX B:

Ammonia Emission Model for Dairy Wastewater Lagoons

Ruihong Zhang, Andrew Chang

Characterization of Dairy Wastewater Lagoons

During spring, summer, and fall of 2002, Deanne Meyer collected samples from the wastewater lagoons of seven representative dairies in the Central Valley. For the sampling, temperature measurements were taken at the surface and below in 1.5 m or 3.0 m increments until reaching the bottom at four locations in each pond. At the same sampling points, liquid manure samples were recovered for measurements of pH (on site) and ammonia (laboratory determination). During sampling, the contents of the ponds are dynamically “boiling” which may have affected the consistency of the sampling protocol. The field data illustrate that the chemical conditions of dairy wastewater lagoon vary with the dairy operations (Table B-1). However, at a given installation, the pH and the total ammonia N concentrations, two of the deciding chemical factors of ammonia volatilization, do not vary much with the season, leaving the climate (temperature and wind velocity) as the critical parameters in deciding the volatilization rates.

Table B-1: *Temperature, pH and ammonia concentration of representative dairy lagoons in the Central Valley.*

Lagoon Dimension (L x W x D)	Date	Temperature °F	pH		Ammonia (mg L ⁻¹)	
			Mean	Standard Deviation	Mean	Standard Deviation
A (730' x 150' x 20')	3/4/02	-	7.65	0.33	884	22
	6/24/02	87	7.34	0.05		
	12/3/02	56	7.37	0.05		
B (1175' x 102' x 20')	3/5/02	-	6.92	0.14	560	21
C (524' x 126' x 20' + 554' x 130' x 20')	3/12/02	-	7.27	0.12	460	24
	3/27/02	-	7.33	0.11		
	3/28/02	-	7.25	0.03		
	3/29/02	-	7.47	0.04		
	6/17/02	-	7.34	0.07		
	6/18/02	-	7.41	0.13		
E (size unknown)	6/28/02	-	7.82	0.09	-	-
F (size unknown)	7/3/02	-	7.50	0.06	998	450
G (size unknown)	7/12/02	-	7.10	0.29	413	128

Ammonia Emission Model

In our opinion, a process-based mechanistic model is needed to evaluate the interactive nature of various factors that contribute to the volatilization of ammonia from dairy manure. We have investigated the possibility of developing an ammonia mass transfer model that, when the rate constants are appropriately calibrated, will be able to compute the NH₃-N volatilization from liquid manure storage lagoons, corral surfaces, and paved surfaces of flush lanes and free stalls. Using this model, we may evaluate the roles of environmental (climatic) factors, N loading rates (or animal density), chemical/biological conditions of water, and physical dimensions of the storage lagoon on ammonia volatilization.

We have developed a mechanistic model for ammonia volatilization that takes into account the chemistry of ammonia (pH, concentration, Henry's law constant), climatic conditions (temperature and wind velocity), and physics of mass transfer (diffusion and mass transfer coefficients). The mathematical equations used to define the ammonia volatilization process are outlined as follows:

NH₃-N emission from lagoon water, mg m⁻² day⁻¹ is:

$$N_{emission} = KF[TAN]$$

where [TAN] is total NH₃-N concentration in the surface layer of lagoon water, K is the overall mass transfer coefficient, and F is the fraction of TAN that is in the free ammonia form.

The fraction of free NH₃ may be expressed as

$$F = \frac{K_d}{K_d + 10^{-pH}}$$

The dissociation constant, K_d, is defined as

$$K_d = 10^{-\left(0.0897 + \frac{2729}{T_{aq}}\right)}$$

where T_{aq} is the temperature of lagoon water in °K.

The overall mass transfer coefficient K (m sec⁻¹) is defined as:

$$K = \frac{k_{LNH_3} K_H k_{GNH_3}}{K_H k_{GNH_3} + k_{LNH_3}}$$

The K_H is Henry's law constant (dimensionless) and is calculated as a function of water temperature (T_{aq}),

$$K_H = \left[\frac{2.395 \times 10^5}{T_{aq}} \right] e^{\frac{4151}{T_{aq}}}$$

K_{GNH_3} and K_{LNH_3} are mass transfer coefficients (m/s) through gaseous and liquid films at the interface of water and air, respectively, and are related to the diffusivities (m^2/s) of ammonia in air (D_{airNH_3}) and ammonia in water (D_{airH_2O}), and ammonia in water ($D_{waterNH_3}$) and oxygen in water (D_{waterO_2})

$$k_{GNH_3} = k_{GH_2O} \left(\frac{D_{airNH_3}}{D_{airH_2O}} \right)^{0.67} \quad k_{LNH_3} = k_{LO_2} \left(\frac{D_{waterNH_3}}{D_{waterO_2}} \right)^{0.57}$$

$$k_{GH_2O} = 18.568 + 703.61\mu_8 \quad k_{LO_2} = 0.6034e^{0.2361\mu_8}$$

where μ_8 is the wind velocity (m/s) at 8 meters above the water surface. The diffusivities are calculated using the following equations,

$$D_{airNH_3} = \frac{10^{-7} T_a^{1.75} \left[\frac{M_{air} + M_{NH_3}}{M_{air} M_{NH_3}} \right]^{0.5}}{P \left[(\Sigma v)_{air}^{\frac{1}{3}} + (\Sigma v)_{NH_3}^{\frac{1}{3}} \right]^2} \quad D_{airH_2O} = \frac{10^{-7} T_a^{1.75} \left[\frac{M_{air} + M_{H_2O}}{M_{air} M_{H_2O}} \right]^{0.5}}{P \left[(\Sigma v)_{air}^{\frac{1}{3}} + (\Sigma v)_{H_2O}^{\frac{1}{3}} \right]^2}$$

$$D_{waterO_2} = 7.28236 \times 10^{-15} \frac{T_{aq}}{e^{\left(\frac{1622}{T_{aq}} - 12.40581 \right)}} \quad D_{waterNH_3} = 6.14526 \times 10^{-15} \frac{T_{aq}}{e^{\left(\frac{1622}{T_{aq}} - 12.40581 \right)}}$$

where M is the molecular weight (g/mol), v is the molecular volume (m^3/mol), T_a is the air temperature (K), and T_{aq} is the lagoon water temperature (K), and v is the atomic diffusion volume (cm^3/mol).

The input parameters for the emission ammonia model include TAN, pH, T_{aq} , T_a , and μ_8 . The TAN and pH are specific to the wastewater characteristics in the storage and T_{aq} , T_a , and μ_8 are specific to climatic conditions at the farm. All these parameters are dynamic and change over time, therefore, the emission rate changes as well. If feasible, TAN and pH in the storage should be monitored on site so that specific values can be used as input for the models. However, appropriate wastewater sampling and analysis procedures and schedules should be used. Frequency of measurement depends on the variability of TAN and pH in the storage and the time scale used for emission calculation. The T_{aq} , T_a and μ_8 should also be measured at the lagoon site. If onsite measurement is difficult, the values for T_a and μ_8 can be obtained from local weather stations and T_{aq} can be estimated by using an empirical equation shown below for T_{aq} (Stefan and Prued'homme, 1993).

$$T_{aq} = 275.9 + 0.86(T_a - 273)$$

Notations:

K_L	overall mass transfer coefficient of ammonia, cm/h
K_H	Henry's coefficient, dimensionless
$k_{L_{NH_3}}$	mass transfer coefficient of ammonia in liquid phase, cm/h
$k_{G_{NH_3}}$	mass transfer coefficient of ammonia in gas phase, cm/h
$k_{L_{O_2}}$	mass transfer coefficient of oxygen in liquid phase, cm/hr
$k_{G_{H_2O}}$	mass transfer coefficient of water in gas phase, cm/hr
u_8	wind speed at 8m height, m/s
u_z	wind speed at an anemometer height z (m), m/s
z_0	roughness height, m
P	atmospheric pressure, atm
$D_{air_{NH_3}}$	NH ₃ diffusion coefficient in air, m ² /s
$D_{air_{H_2O}}$	H ₂ O diffusion coefficient in air, m ² /s
$D_{water_{O_2}}$	O ₂ diffusion coefficient in water, m ² /s
$D_{water_{NH_3}}$	NH ₃ diffusion coefficient in water, m ² /s
M_{air}	molecular weight of air, g/mol (29)
M_{NH_3}	molecular weight of NH ₃ , g/mol (17)
M_{H_2O}	molecular weight of H ₂ O, g/mol (18)
$(\Sigma v)_{air}$	air atomic diffusion volume, 20.1 cm ³ /mol
$(\Sigma v)_{NH_3}$	NH ₃ atomic diffusion volume, 14.9 cm ³ /mol
T_a	air temperature, K
T_{aq}	manure temperature, K

To demonstrate the ammonia emission model, ammonia emission rate from a dairy wastewater lagoon for a 1000 hd dairy was calculated with assumed levels for pH (7.0, 7.4, 7.8), surface temperature (ambient temperature), ammonia concentration (300, 450, 600 mg/L), and lagoon depth (10, 25 ft). The wastewater volume generated by the dairy was assumed to be 100 gal/hd/day. Using an average nitrogen excretion of 460 g head⁻¹ day⁻¹, total nitrogen concentration in fresh wastewater is about 1200 mg L⁻¹. Assuming three-month storage, this lagoon has a volume of about 9 million gallons. Results of the sample computations for the dairy waste water lagoon in Fresno are summarized in the following Table B-2.

Table B-2: Ammonia emission calculations

Lagoon Conditions			NH ₃ -N Emission (kg head ⁻¹ yr ⁻¹) ⁺
Depth, m (ft)	pH	NH ₃ -N, mg l ⁻¹	
7.62 (25)	7.0	300	3.0 (6.6)
		450	4.4 (9.7)
		600	5.9 (12.9)
	7.4	300	7.4 (16.3)
		450	11.1 (24.4)
		600	14.8 (32.6)
	7.8	300	18.0 (39.6)
		450	27.3 (60.1)
		600	36.4 (80.2)
3.05 (10)	7.0	300	5.1 (11.2)
		450	7.6 (16.7)
		600	10.2 (22.5)
	7.4	300	12.7 (27.9)
		450	19.0 (41.9)
		600	25.3 (55.7)
	7.8	300	31.2 (68.7)
		450	46.8 (103.1)
		600	62.4 (137.4)

⁺Values in the parenthesis denotes NH₃-N emission in pounds per head of animal per year.

The graphs in the following pages (Figures B-1 to B-5) illustrate that factors, such as depth of the lagoon, pH, concentration of ammonia in the lagoon water, and location (Fresno vs. San Joaquin) significantly affect the monthly ammonia volatilization from wastewater storage lagoons. Generally, low temperature, low pH (pH = 7 or less), high water level (i.e. 25 ft), and low ammonia concentration (i.e. 300 mg L⁻¹) reduce the ammonia volatilization.

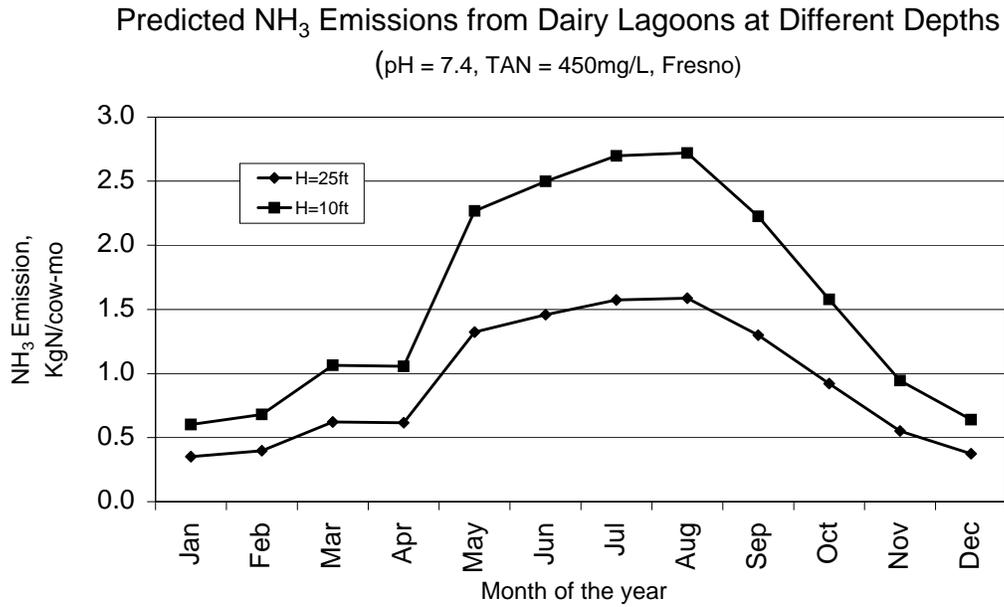


Figure B-1: Simulated ammonia loss in different months of the year from dairy wastewater storage lagoon (Fresno), pond depth 10 ft vs. 25 ft. TAN is total NH₃-N concentration in the surface layer of lagoon water.

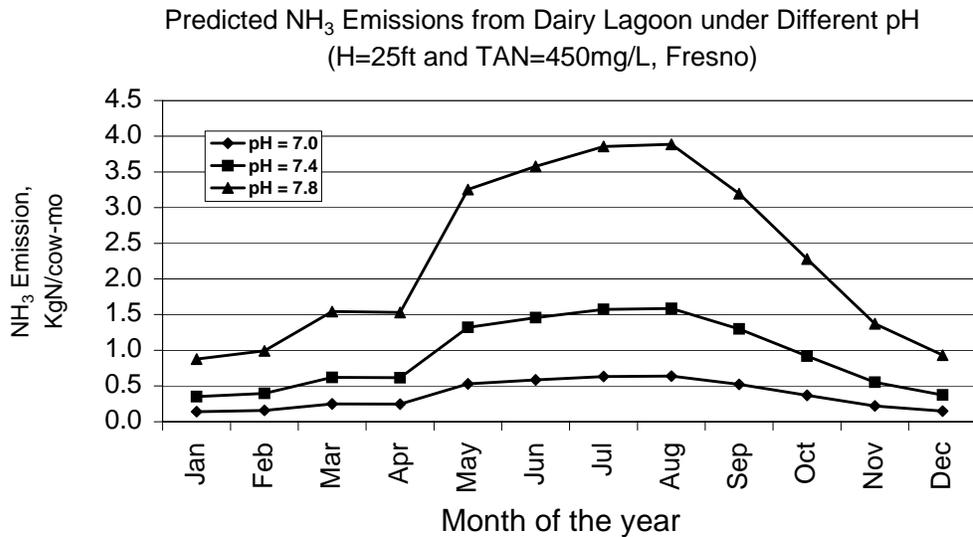


Figure B-2: Simulated monthly variation of ammonia losses from a dairy wastewater storage lagoon (Fresno climate) at pH 7.0, 7.4, and 7.8. TAN is total NH₃-N concentration in the surface layer of lagoon water.

Predicted NH₃ Emissions under Different NH₃-N Concentrations

(H = 25ft, pH = 7.4, Fresno)

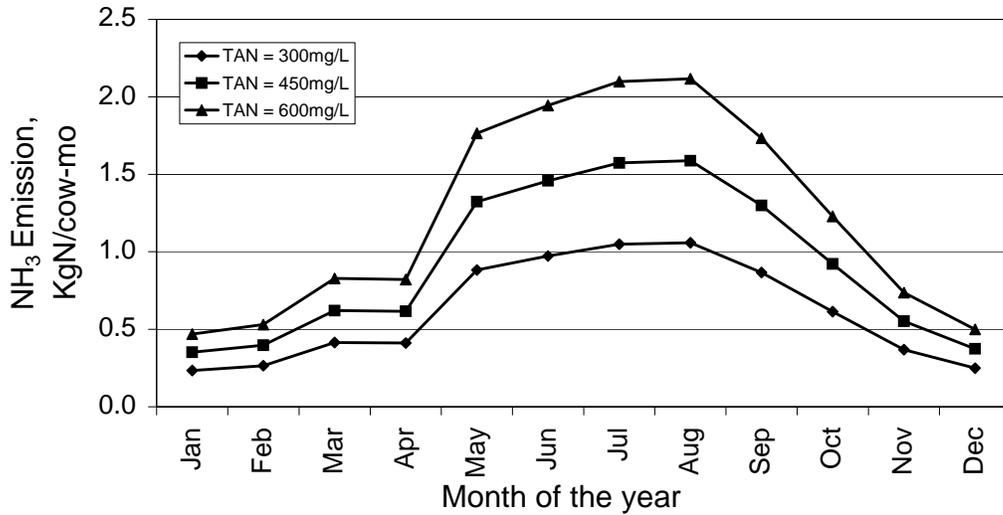


Figure B-3: Simulated monthly ammonia losses from a dairy wastewater storage lagoon (Fresno climate) for total ammonia nitrogen (TAN) concentrations of 300, 450, and 600 mg L⁻¹

Predicted NH₃ Emissions from Dairy Lagoons at Extreme Conditions

(Fresno)

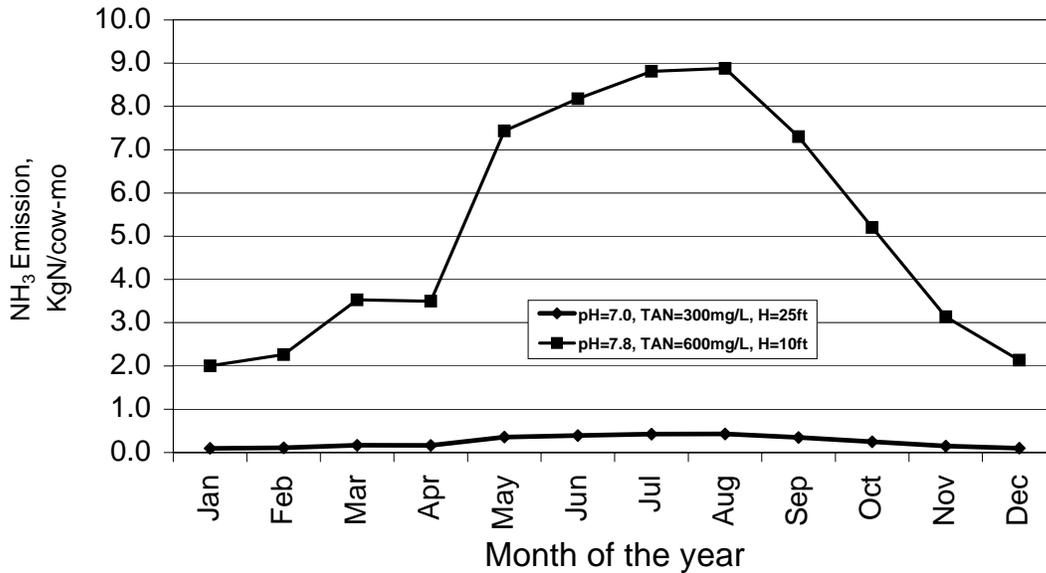


Figure B-4: Simulated monthly ammonia losses from a dairy wastewater storage lagoon (Fresno climate) under worst (pH = 7.8, H = 10 ft, TAN = 600 mg l⁻¹) and best (pH = 7.0, H = 25 ft, TAN = 300 mg l⁻¹) environmental conditions.

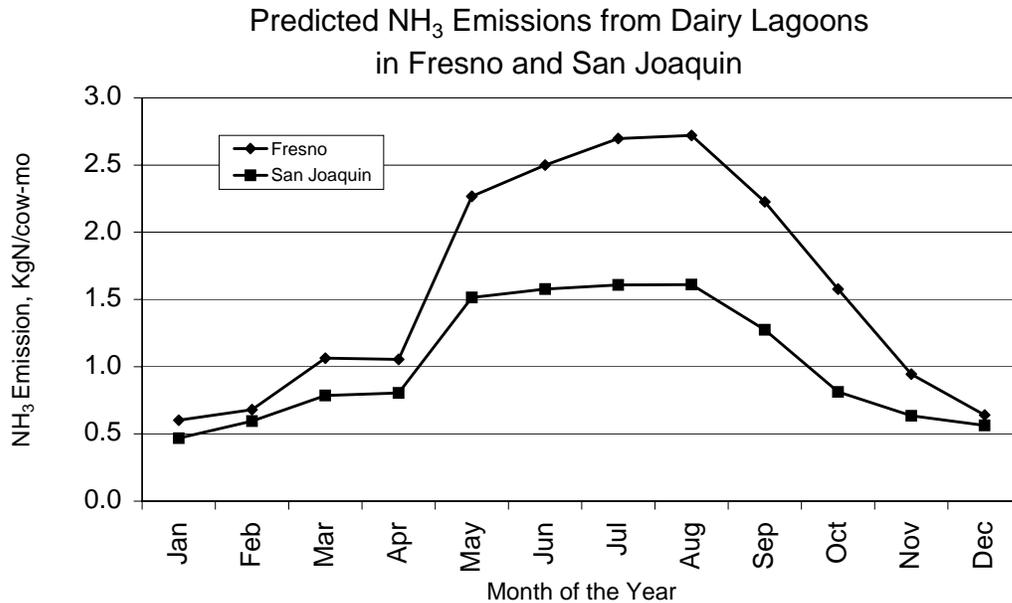


Figure B-5: Simulated monthly ammonia losses from a dairy wastewater storage lagoon: comparison for Fresno vs. San Joaquin climate data (pH = 7.4, TAN = 450 mg l⁻¹, and depth = 10 ft).

The graphs of Figures B-1 through B-5 may be integrated to determine the annual ammonia volatilization loss from the lagoons. To illustrate the range at which the ammonia volatilization may take place, the best and the worst possible scenarios calculate out to be 11 and 63 kg of ammonia volatilization head⁻¹ yr⁻¹ (Figure B-5). The estimated values are similar in magnitude to those derived from field measurements as reported in the literature.

Ammonia Emission Model Validation

The mass transfer model has been validated by conducting field ammonia volatilization experiments under controlled conditions (Figure B-6). The volatilization of ammonia from clean water and dairy wastewater were measured over a one-week period of time. The environmental conditions (water temperature, air temperature, relative humidity, and wind velocity) were continuously recorded throughout the experimental period (Figures B-7 and B-8).

In general, there were diurnal variations in the environmental conditions. The relative humidity rose continuously over the evening hours and reached a peak between 6 a.m. and 7 a.m. The temperatures of clean and wastewater were similar and were at their height between 1 p.m. and 8 p.m. They gradually cooled off during the evening hours and then started rising around 7 a.m. The wind velocity varied considerably. The wind was calm in the morning hours, started to pick up speed in the mid-afternoon, and reached a peak at approximately 6 p.m. to 7 p.m. There were considerably fewer day-to-day changes in the environmental conditions.



Figure B-6: *Ammonia volatilization experiments for model validation*

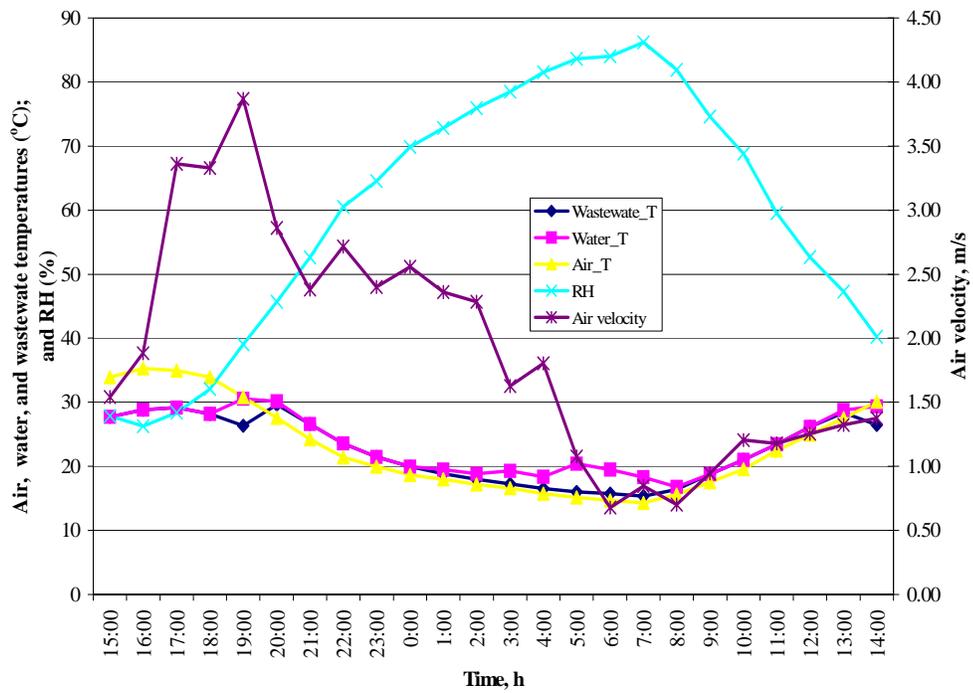


Figure B-7: *Variations of the environmental factors during a typical summer day showing a diurnal cycle*

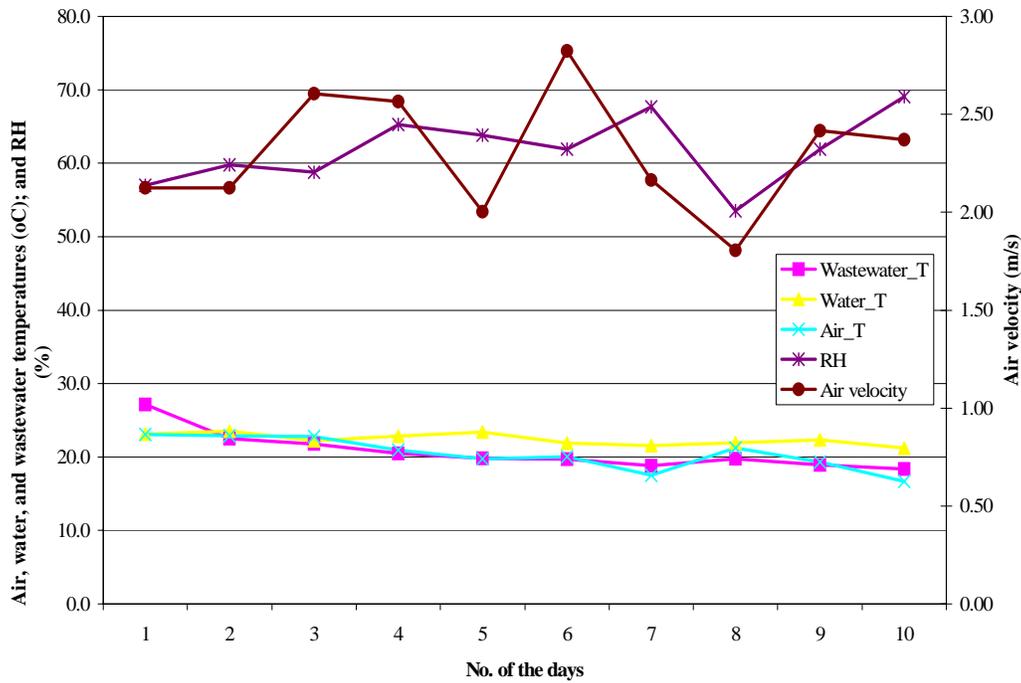


Figure B-8: Average daily measurements of the environmental conditions during the ammonia volatilization experiments

The recorded measurements of the environmental conditions were used as inputs for the model to simulate the ammonia volatilizations from the clean and from the wastewater. The simulated and the measured daily ammonia volatilization rates were compared (Table B-3). The discrepancies between the measured and predicted values are within 20% of measured values.

Table B-3: Simulated and measured ammonia volatilization of clean water

Day	pH	TAN mg/L	Water		pH	TAN mg/L	Wastewater	
			Measured loss g/m ² .d	Predicted loss g/m ² .d			Measured loss g/m ² .d	Predicted loss g/m ² .d
0	9.02	429			9.20	499		
1	8.89	440	11.96	14.34	9.17	447	22.43	18.95
2	8.73	420	6.64	8.21	9.15	409	15.60	15.60
4	8.56	381	4.93	4.03	9.12	327	11.31	11.47
6	8.40	368	1.93	2.35	9.12	264	6.26	8.04
7	8.26	370	2.84	1.53	9.04	251	5.13	5.95
8	8.13	371	1.95	1.55	9.01	242	5.03	6.34
9	7.98	374	1.51	1.20	9.03	233	6.54	6.52

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APPENDIX C: Illustration of Nitrogen Leaching Losses in Manure Systems Subject to Organic Nitrogen Mineralization

Andrew Chang

Nitrate leaching in cropland soils is the result of dynamic processes that involve organic N mineralization, plant uptake of N, irrigation scheduling and plant stresses due to water and/or N deficiencies. The dynamic interrelations between N inputs, N uptake, and N leaching can be captured in model simulations. Here we used the computer model ENVIRO-GRO (Pang and Letey, 2000) to illustrate nitrogen leaching losses in forage crops grown on manure fertilizer only. In the simulations, manure nutrients were applied at specified nutrient levels throughout the crop growing season at bi-weekly intervals. Half of the applied N was assumed to be in organic nitrogen form, half was assumed to be in the ammonia form. Details of the simulations are given in Appendix E, section 2.

If the N inputs from manure equal the 100% maximum N crop uptake potential, a 100% relative yield would only be achieved if leaching and other nitrate losses were eliminated. In reality some leaching and other losses occur. Hence, the crop will be stressed, relative yields and evapotranspiration decline, crop N uptake will be lower, and even more nitrate will be available for leaching (Figure C-1). The N inputs exhibit the most significant impact on nitrate leaching. Leachable nitrate levels in the soil profile initially increase over time and approach a steady-state condition within approximately five years (Figures C-2 to C-4). The simulations demonstrate that 100% relative crop yields may be achieved only if the N input levels are significantly greater than the maximum crop N uptake potential (i.e., 117, 163, and 231% of the maximum crop N uptake potentials considered in the simulations).

When N input is at 117% of the maximum crop N uptake potential, the nitrate leaching potential of the soils is well controlled and approaches a steady-state condition within 2-3 years. At the low leaching fraction (i.e. $LF = 0.07$), N is expected to accumulate in the soil profile as nitrate leaching remains low and steady over time. With more realistic leaching fractions (i.e. $LF = 0.18$ to 0.29), essentially all excess N is expected to be mineralized and is potentially leachable.

At higher N input levels (163 and 231% of maximum crop N uptake potentials), excess N is expected to be released as nitrate. The time required to reach the steady-state equilibrium is dependent on the leaching fractions. At the low leaching fraction ($LF = 0.07$), the nitrate leaching potential rises slowly and is expected to take a relatively long time to reach steady-state. At higher, more realistic leaching fractions, however, the nitrate leaching potential will rise rapidly over time and reaches steady-state conditions in approximately three simulated years (Figure C-4).

On fields that have been receiving the same amount of manure annually for five or more years, the soil organic N level is likely to be in a steady-state. Nitrate is generated from the organic N pool at an approximately constant rate as illustrated in the model

simulations. In such a situation, and if inorganic N applications are properly timed, the percent of nitrate leached is likely to closely track the irrigation leaching percentage. Hanson et al. (1999) found that a leaching fraction of 15% to 30% is practically achievable in border check and furrow systems. Hence, we can expect that the nitrate leaching, under the best of field conditions, is approximately on the order of 10% to 15% of the N applied.

Some field studies have shown that in irrigated farming, the mass of nitrate leached beyond the crop root zone is related to the volume of water that percolates beyond the root zone. Rible et al. (1979) took deep soil cores at 58 sites in California under a wide range of cropped and non-cropped systems. Drainage volume at the sites was estimated to range from 1 to 41 inches per year. The authors determined an approximate relationship between mass of N leached and volume of deep percolation, with 3.4 lb nitrate-N ac⁻¹ yr⁻¹ (3.8 kg nitrate N ha⁻¹ yr⁻¹) leached for each inch (2.54 cm) of water draining below the root zone in the same time period. This corresponds to a concentration of nitrate-N in the drainage water of approximately 15 mg L⁻¹. Rible et al. (1979) observed no significant relationship between the mass of nitrate-N leached and the amount of N applied to the land, probably due to the wide range of land use, vegetation, and management across the 58 sites. Drainage volume was a far more important controlling factor.

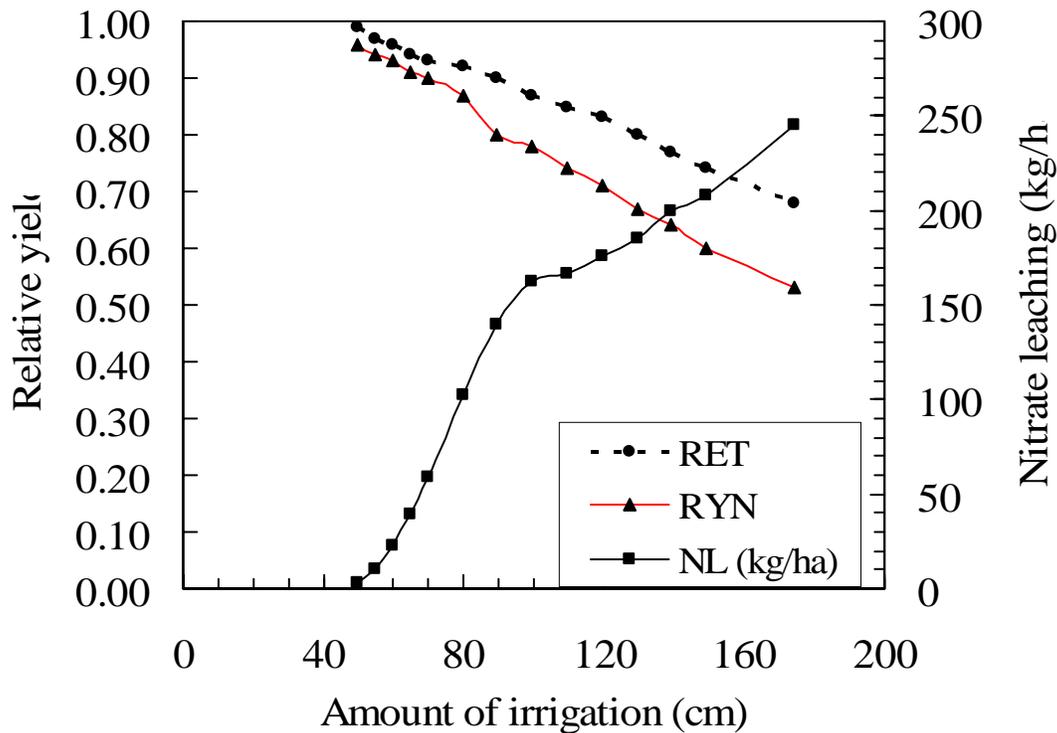


Figure C-1: Simulated relative yields (RYN), relative evapotranspiration (RET), and nitrate leaching (NL) at various irrigation amounts. N input is equal to 100% of the maximum crop N uptake potential.

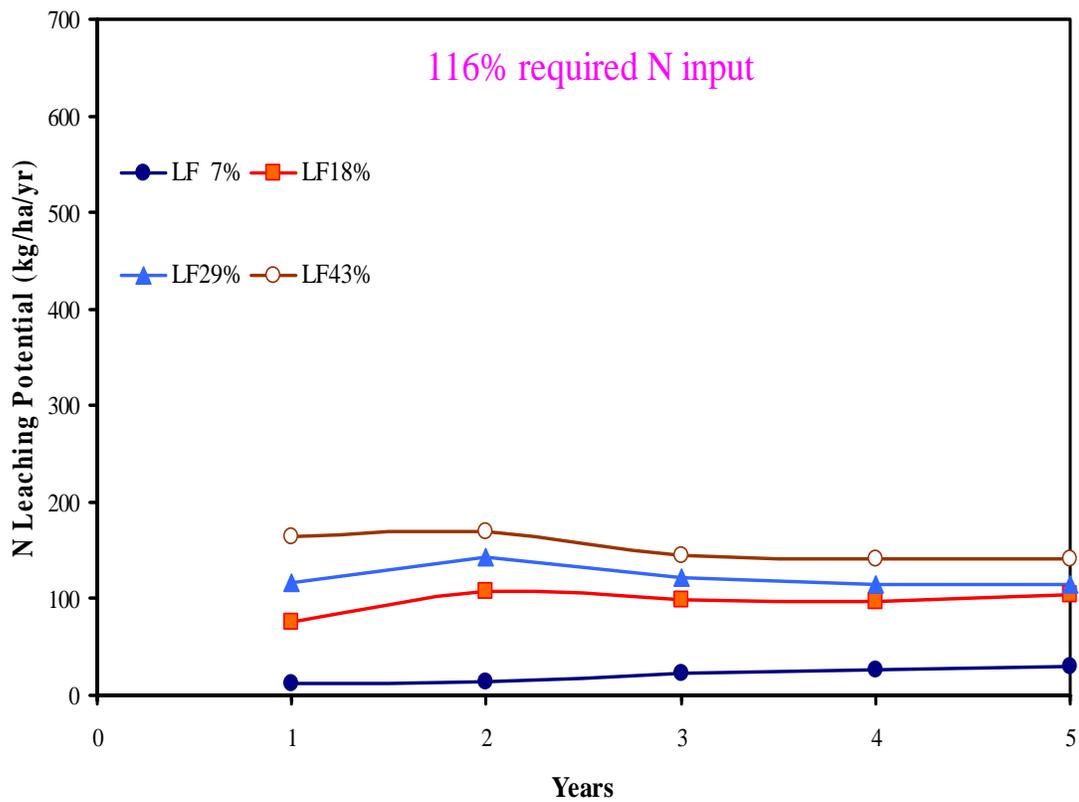


Figure C-2: Simulated potential N leaching loss at four different irrigation regimes for year round forage production that receives N applications equivalent to 117% of the maximum crop N uptake of $600 \text{ kg ha}^{-1} \text{ yr}^{-1}$. The leaching fraction (LF) of the four irrigation regimes range from 7% to 43%.

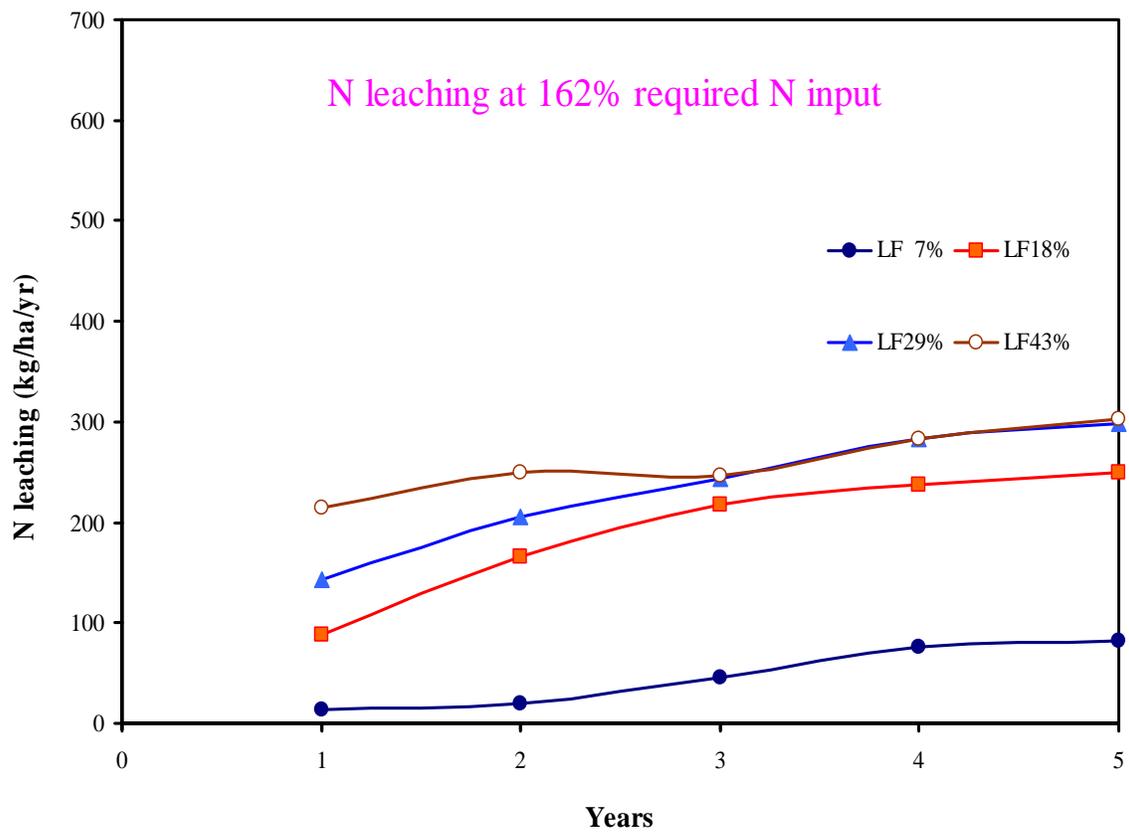


Figure C-3: Simulated potential N leaching loss at four different irrigation regimes for year round forage production that receives N applications equivalent to 162% of the maximum crop N uptake of $600 \text{ kg ha}^{-1} \text{ yr}^{-1}$. The leaching fraction (LF) of the four irrigation regimes range from 7% to 43%.

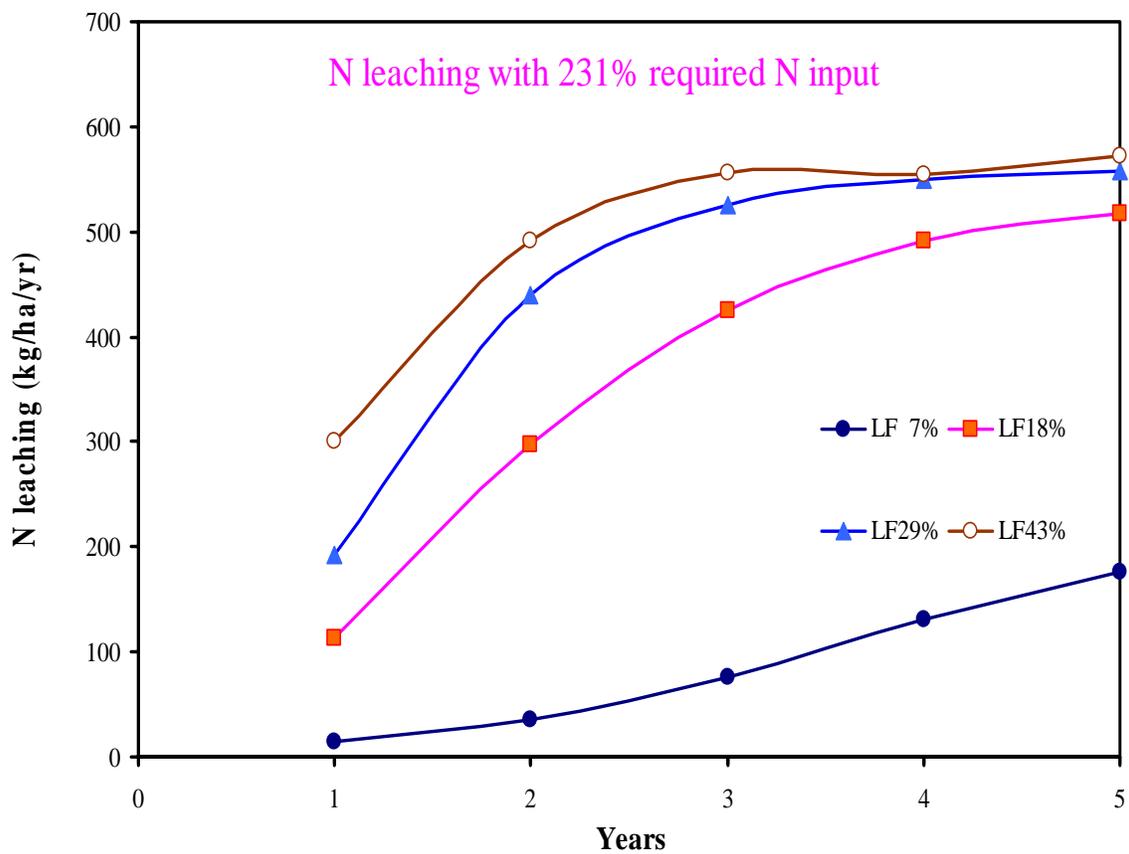


Figure C-4: Simulated potential N leaching loss at four different irrigation regimes for year round forage production that receives N applications equivalent to 231% of the maximum crop N uptake of $600 \text{ kg ha}^{-1} \text{ yr}^{-1}$. The leaching fractions (LF) of the four irrigation regimes range from 7% to 43%.

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APPENDIX D: Mineralization – Conceptual Model and Field Studies

Andrew Chang

Perhaps the most significant difference between the use of commercial fertilizer and manure in nutrient management is the significant amount of organic nitrogen present in the manure. Organic nitrogen that is applied to soils during land application is not immediately available for plant uptake. Rather, it is subject to natural biochemical degradation processes that are generally referred to as “mineralization”. Ultimately the degradation processes yield nitrogen in inorganic form (ammonium). Only in the inorganic form is nitrogen available to plants. For purposes of nutrient management, the amount of organic nitrogen that ultimately mineralizes and the rate at which organic nitrogen is mineralized and, hence, available for plant uptake, is important. Temperature, but also soil moisture, soil organic matter content, and the age of the manure have significant influence on the mineralization rate. Here we provide an overview of the conceptual models for mineralization and a brief review of the literature on mineralization of organic nitrogen applied with manure.

Mathematical Expression. Salter and Green (1933) demonstrated that soil organic matter and nitrogen transformations could be described by first-order reaction kinetics. Stanford and Smith (1972) expressed the organic N mineralization with the equation:

$$N_m(t) = N_o \times (1 - e^{-\lambda t}) \quad [\text{Eq. D-1}]$$

where $N_m(t)$ (mg kg^{-1}) is the mineralized N at time t (days), N_o (mg kg^{-1}) is the mineralization potential (total mineralizable organic N), and λ (d^{-1}) is the first-order reaction N mineralization rate constant. Parameter estimation has been made by a nonlinear least squares method (Smith et al. 1986; Deans et al., 1986). Representative values for net mineralization rates and first-order rate constants for some ecosystems have been published by Smith and Paul (1990).

Deans et al. (1986) proposed two models for N mineralization; the first model took the mathematical form of Equation D-1 and the second used a double exponential equation:

$$N_m(t) = N_o \times S \times (1 - e^{-ht}) + N_o \times (1-S) \times (1 - e^{-kt}) \quad [\text{Eq. D-2}]$$

where S and $(1-S)$ represent labile and recalcitrant organic N fractions decomposing at rates h and k , respectively. Their work indicated that the double exponential equation resulted in a better fit of experimental data on N mineralization. Cabrera and Kissel (1988) showed that the rate constants k and h and the apparent size of the two nitrogen pools, S and $(1-S)$ were significantly affected by the length of the incubation time.

Paustain et al. (1992) modeled as many as five pools of decomposing organic matter, which they describe as plant residue broken into structural and metabolic pools, and soil organic matter (SOM) separated into three pools -- active, slow and passive. Others may

have created as many as seven separate organic nitrogen pools to model the decomposition of SOM (Norton, 2000). With limited data points, the divisions of SOM and the associated nitrogen into multiple pools resulted in better regression models. The subdivisions reflected little reality in terms of the nature of SOM and many of the pools were not physically definable.

Pratt et al. (1973) presented an approach for calculating the annual rates of N mineralization expressed as a series of fractional proportions of any given application of manure, hereinafter referred to as a decay series. For fresh bovine manure with 3.5% N dry weight, they proposed a 0.75, 0.15, 0.10, 0.05 decay series, meaning that if 100 kg of manure N were applied in this form, 75% of the N would be available in mineralized form during the first year (75 kg); 15% of the remaining 25 kg would mineralize the second year (4 kg); 10% of the remaining 21 kg would mineralize the third year (2 kg); and so on. Subsequent releases were thought to be negligible. The amount of N available in the first year is much larger because it includes the N already in mineral form, and organic N that is readily mineralizable.

Pratt et al. (1973) determined that much less N would mineralize the first year if the manure had been exposed over time to the weather while deposited and dried outdoors before collection for disposal. Consequently, the total N contents of the dried and weathered dairy manure are considerably lower. Nitrogen decay series for these manure types vary with the total N contents and their calculated λ and half-life values are given in Table D-1.

Table D-1: *N Mineralization for different types of bovine manure*¹

Decay series	Manure Type	λ (day ⁻¹)	Half-life (day)
0.75, 0.15, 0.10, 0.05	Fresh 3.5% N	0.0038	180
0.40, 0.25, 0.06	Dried 2.5% N	0.0014	500
0.35, 0.15, 0.10, 0.05	Dried 1.5 % N	0.0012	590
0.20, 0.10, 0.05	Dried 1.0% N	0.00061	1100

¹Values of λ and half-life were calculated from the first year N releases of the decay series using Eq. 6-1.

N Transformations of Dairy Manure. We obtained data from field studies where dairy manure had been applied, crops grown, and the mineralization of N was either determined or could be estimated from crop yields such that λ values for Eq. D-1 could be calculated. Studies which include all of the above factors are very rare. There seems to be abundant data for N mineralization in natural ecosystems, but studies where dairy manure was applied are limited.

Pratt et al. (1976ab) report a study where dairy manure and liquid steer manure were applied at various rates over a four-year period at the Moreno Field Station of the University of California, Riverside. The four-year field trial tracked the yields of Sudan grass and barley; quantities of N accumulated in the crop dry matter and in the soil organic matter; losses due to nitrate leaching; and annual amounts of N applied. Results

indicated that 40 to 50% of applied N mineralized during the first year and 10 to 20% of the remainder mineralized the second year. These figures would provide λ values and half lives comparable to the range presented in Table D-4. The measured total N mineralization for the four-year period were within 15% of the calculated N releases according to Table D-4.

Talarczyk et al. (1996) report a 1992 to 1995 study of corn grown with dairy manure in Wisconsin. Fresh dairy manure tested for N, P, K, and sulfur was applied at a uniform rate of 35 tons per acre (78.6 mT ha^{-1}) on separate plots in November of the previous year, January and March without incorporation until April, and on plots for a fourth treatment in April with incorporation 4 to 12 days after application. The results of these treatments were compared with a check plot and plots that received pre-plant urea fertilizer at rates of 75, 125, and 175 pounds N per acre ($84, 150, 196 \text{ kg ha}^{-1}$). All plots received a starter fertilizer of 8-32-17 applied in a band at planting. Each treatment was replicated three times in randomized blocks.

As the November, January, and March applications remained on the soil surface for a long time prior to incorporation, there were opportunities for N to be lost prior to the incorporation. This would cause uncertainties in assessing the N transformations. There may also have been leaching losses that have not been accounted for. Assuming that the quantity of N available to the crop from the manure came from mineralization and that there were no losses of mineralized N via denitrification or leaching, we may calculate a lower bound for k, the first order reaction rate constant, accordingly. First, the yields of corn for grain using the four-urea application rates were plotted and equations obtained for yield vs. N application. Using these regression equations and the measured yield for the manure plots, the mineralized N was calculated. Since the total N in the 35 tons of manure was known and the mineralized N was estimated from the yields, Equation 1 could be applied to calculate the lowest possible value of k. The N release of the April manure application may be used to assess the N mineralization rate for the summer. The N releases of November and April manure applications could then be compared to determine the mineralization rate for the winter.

Although the authors state that no manure was applied to the plots for two years prior to the beginning of this experiment in 1992, they do not seem to account for N that might be mineralized in the second, third and fourth years of the experiment. It is understood that 35 tons per acre (78.6 mT ha^{-1}) were applied each year. While the data was not perfect, it offered an opportunity to compare the N mineralization of cold (November to April) vs. warm (April to October) seasons. Table D-2 reports the results of our calculations.

Table D-2: Calculation of λ values and half-life for Wisconsin plots. Due to the assumptions made in the calculation (no significant losses due to volatilization, denitrification, or leaching), these values are only minimum (λ values) and maximum (half-life) values, respectively.

Description	1992	1993	1994
1. Manure applied (tons wet manure per acre)	35.0	35.0	35.0
2. Mean value total N (lbs per ton wet manure)	12.6	11.3	10.2
3. Total N applied (lb ac ⁻¹)	441	396	357
4. November manure Corn yield (bushels per acre)	176	114	179
5. April manure Corn yield (bushels per acre)	171	94	170
6. November N mineralized (lbs ac ⁻¹)	135	74	225
7. April N mineralized (lbs ac ⁻¹)	102	16	69
8. Winter mineralization (lbs ac ⁻¹) = $\Delta(6 - 7)$	33	58	156
9. November – April λ value (day ⁻¹)	0.00052	0.00025	0.0012
10. November - April half-life (day)	1350	2770	542
11. April - October λ value (day ⁻¹)	0.0016	0.0011	0.0038
12. April - October half-life (day)	443	660	182

The first finding, which is readily apparent, is the seasonal (i.e. temperature) effect on the mineralization rates. The rates are much lower in winter and early spring months than during the growing season. The authors report that 1993 had a cold wet spring and much moisture into June and July. One might anticipate denitrification under these conditions, but the greater yield for the November manure application would seem to indicate that the problem was lack of N mineralization. The authors estimated that only 75 lbs ac⁻¹ (84 kg ha⁻¹) was available from the November manure applications and only 40 lbs ac⁻¹ (45 kg ha⁻¹) for the winter and spring manure applications. Our figures indicate that even less N was available for the April manure application in 1993.

In all, the N mineralization rates indicated by the data from the Wisconsin study are probably low because additional N was likely mineralized and either leached as nitrate or denitrified as nitrogen gases to the atmosphere. Nevertheless, the λ values calculated for the growing season (i.e. the April manure application) certainly fall within the range of those calculated from the Pratt et al. (1973) decay series (0.00061 to 0.0038 day⁻¹). The effect of temperature is evident by the N mineralization rates of cold (Nov – April) and warm (April through October) seasons as well as the problems encountered in the years 1993 and 1995 with cold wet springs and early summers. In addition to soil temperature, the other factors highlighted in the literature as influencing microbial activity and mineralization are soil moisture, pH, characteristics of the substrate, and management practices.

A study from Stephenville, Texas (Haney et al., 1996) was also used for comparison of λ values. Dairy manure, not described further, was applied at rates of 0, 75, 150, and 300 lbs N per acre (84, 168, 337 kg ha⁻¹) on plots used either to grow coastal Bermuda grass or a dual crop of wheat and coastal Bermuda grass. Considering the similarities in climatic conditions and dairy operations in Texas and California, the dairy manure in Texas would be comparable to solid dairy manure collected in Central Valley of California. The investigators measured the initial inorganic N in the soil, the total N uptake by the crops, and the N mineralized. The N mineralized may be determined by mass balance that included the initial soil N, the N from the manure, and N uptake by plants. Thus, total N mineralized from dairy manure treated plots minus the N mineralized in the control plots equals the N mineralized from manure. Haney et al. (1996) results are reproduced in Table D-3 and the resulting λ values and N mineralization half-life are summarized in Table D-4.

Table D-3: *N mass balance for dairy manure applied for forage production in Stephenville, Texas (Haney et al., 1996)*

Cropping system	Manure N added (lb ac ⁻¹)	Initial inorganic N (lb ac ⁻¹)	Total N uptake (lb ac ⁻¹)	Total N mineralized (lb ac ⁻¹)	Manure N mineralized (lb ac ⁻¹)	Manure N utilized (%)
Wheat + Coastal Bermuda Grass	0	20	45	25	0	-
	75	21	78	57	32	43
	150	21	116	95	70	46
	300	32	173	141	116	38
Coastal Bermuda Grass Only	0	23	58	35	0	-
	75	32	95	63	28	37
	150	42	136	94	59	39
	300	36	188	152	117	39

Table D-4: *Calculated λ values and half-life according to Haney et al. (1996) data in Table D-3*

Cropping system	Manure N added (lb ac ⁻¹)	Manure N mineralized (lb ac ⁻¹)	Value of λ^* (day ⁻¹)	Half-life* (day)
Wheat + Coastal Bermuda grass	0	0	-	-
	75	32	0.0025	280
	150	70	0.0028	248
	300	116	0.0022	318
Coastal Bermuda grass Only	0	0	-	-
	75	28	0.0021	334
	150	59	0.0022	312
	300	117	0.0022	315

* Calculated using Equation 1, the data from Table 3 and 225-day growing season.

Again, assessment of the data from the Texas study show that the λ values calculated for the growing season certainly fall within the range of those calculated from the Pratt et al. (1973) decay series (0.00061 to 0.0038 day⁻¹). Using Haney et al. (1996) data for total N mineralized there seems to be a pattern of decreasing N mineralization rates as the total N in the manure increases. This is logical as the microbial decomposition processes may be affected by factors other than amounts of substrates. Excessive application of manure may simply overload the system and not provide additional available N to the crops.

Van Kessel and Reeves (2002) represents perhaps the most complete recent study on the mineralization of nitrogen in dairy manure. They collected 107 samples of manure from five eastern states, which had wide ranges of composition: 14 to 386 g dry matter kg⁻¹; 0.9 to 9.5 kg total N m⁻³; and 0.3 to 4.9 kg NH₄⁺-N m⁻³. These samples were incubated aerobically for 56 days at 25 °C and sampled for NH₄⁺-N and NO₃⁻-N on days 2 and 56. Net N mineralization ranged from -29.2% (i.e. immobilization of NH₄⁺-N) to +54.9% of total N. There appeared to be no simple straightforward relationship between the composition of manure and the extent of N mineralization. Even though there is no significant correlated relationship with the components, one of the better r-values obtained was the C/N ratio with r =0.348. The histogram of the results appears to provide a normal relationship with an average mineralization of 12.8%. Calculation of λ values and half-life from this provides $\lambda = 0.0024$ day⁻¹ with a half-life of 283 days - easily comparable to the results from the field studies presented earlier. The range in half-life is from infinite to 49 days.

Van Kessel et al. (2000) provided data for the mineralization of components found in manure. Calculations for the λ values and half-life of these components are presented in Table 6-5 below. One may observe a close but not perfect relationship between the C to N ratio and the half-life for nitrogen mineralization. The values for neutral detergent fiber (NDF) and acid detergent fiber (ADF) were not calculated since there was a net immobilization of N for these components. One might expect this from material that had high C/N ratio.

Table D-5: *Values for λ and half-life of various manure components*

Component	C/N ratio	λ (day⁻¹)	Half-life (days)
Urea	0.4	1.33	0.52
Trypticase	3.4	0.012	58
Immature alfalfa	10.7	0.0064	109
Mature alfalfa	14.9	0.0046	152
Immature orchard grass	14.2	0.0047	147
Mature orchard grass	22.3	0.0012	557
NDF/ADF ⁺	42/145	Not calculated	Not calculated
Soybean meal	5.4	0.0091	76
Roasted soybean meal	7.7	0.0057	122
Bacteria	4.5	0.0099	70
Colonic cells	4.3	0.0096	72

⁺NDF and ADF refer to non-digestible fiber and acid digestible fiber, respectively.

Nakamura and Harter (Appendix H) used field data from a study in Merced County (Harter et al., 2001; Campbell-Mathews, 2004). They constructed a site-specific fully transient unsaturated zone flow and transport model that was calibrated against a three-year time series of soil moisture and soil ammonia measurements and against groundwater nitrate measurements. The program HYDRUS (Šimůnek et al., 1998) was used for the modeling. It operates similar to ENVIRO-GRO (Appendix E), except that it does not allow for simulation of plant stress conditions. Transient water flow and nitrogen transformation and transport was modeled. The model also accounted for plant nitrogen uptake assuming that the crop was neither water nor nutrient stressed. Nitrogen transformations were handled as described above and also accounted for the sorption of organic N and ammonia-N.

The model was used to estimate mineralization, nitrification, and denitrification rates by calibration against a large set of soil nitrogen and groundwater nitrate data. A sensitivity analysis was implemented to investigate the sensitivity of the estimated parameters to the model structure. The results confirmed that denitrification and/or volatilization from the root zone and even during the application of the diluted manure water was negligible at the site and that nitrification of ammonia is relatively rapid (less than 2-3 months half-life) and affected all of the organic manure. This apparently rapid mineralization of organic N may be partly due to high permeability and hence good aeration of the fine sandy textured soils at the field site. It is possibly due also to mineralization of residual organic N from manure applications in years prior to the experiment, which had seen significantly higher amounts of organic N applications. Several other studies have shown cases in which manure led to significant denitrification (Calderon et al., 2004).

With respect to mineralization rates (eq. D-1), the analysis by Nakamura and Harter (Appendix H) also indicates that mineralization of the applied organic manure was relatively rapid (less than 2-3 months half-life) and affected all of the organic manure (all

manure organic N was found labile). This apparently rapid mineralization of organic N may be partly due to high permeability and hence good aeration of the fine sandy textured soils at the field site. It is possibly also due to mineralization of residual organic N from manure applications in years prior to the experiment, which had seen significantly higher amounts of organic N applications.

In support of complete, relatively rapid mineralization under California conditions is a recent review of biosolids N mineralization by Crohn (Appendix I). Biosolids (human waste from wastewater treatment plants) are essentially similar in organic N composition to animal manure. Hence, the mineralization rates of organic N in animal manure would be expected to be of similar order of magnitude as that of organic N in biosolids.

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APPENDIX E:

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Simulating Dairy Liquid Waste Management Options as a Nitrogen Source for Crops

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Abstract

Large scale dairy operations are common. In many cases the manure is deposited on a paved surface and then removed with a flushing system, after which the solids are separated, the liquid stored in ponds, and eventually the liquid applied on adjacent crop land. Management of liquid manure to maximize the fertilizer value and minimize water quality degradation requires knowledge of the interactive effects of mineralization of organic N (ON) to NH_4^+ , crop uptake of mineral N, and leaching of NO_3^- on a temporal basis. The purpose of the research was to use the ENVIRO-GRO model to simulate how the amount of applied N, timing of N application, ON mineralization rates, chemical form of N applied, and irrigation uniformity affected (1) yields of corn (*Zea mays*) in summer and a forage grass in winter in a Mediterranean climate and (2) the amount of NO_3^- leached below the root zone. This management practice is typical for dairies in the San Joaquin Valley of California. The simulations were conducted for a ten-year period. Steady state conditions, whereby an equivalent amount of N applied in the organic form will be mineralized in a given year, are achieved more rapidly for materials with high mineralization rates. Both timing and total quantity of N application are important in affecting crop yield and potential N leaching. Major conclusions from the simulations are as follows. Frequent low applications are preferred to less frequent higher applications. Increasing the amount of N application increased both the crop yield and the amount of NO_3^- leached. Increasing irrigation uniformity increased crop yields but had variable effects on the amount of NO_3^- leached. A winter forage crop following a summer corn crop effectively reduced the leaching of residual soil N following the corn crop.

Key words: Nitrate, manure, groundwater, organic nitrogen, mineralization, leaching.

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E.1 Introduction

Livestock and dairy production around the world is progressively moving toward congregating a large number of animals into small land areas. For example, dairies in California have a total herd size of 1.5 million cows. In 1999 the average size of California's 2200 dairy farms was over 650 milk cows, not including dry stocks, heifers, and calves (CDFA, 2000).

The feed rations for animals in the confined animal operations are formulated to maximize production. As a result, the large amount of nitrogen-rich wastes produced by the animals must be properly managed to avoid environmental degradation.

In the San Joaquin Valley of California, manure deposited on paved surfaces in dairies is removed with a flushing system. After separating out solids, the liquid manure is typically stored in ponds (lagoons) and eventually applied to adjacent cropland. Liquid manure applied to crop land serves as a fertilizer nutrient source for crops and may become a potential source of nitrate (NO_3^-) groundwater degradation if the land applications are not properly managed.

Forage crops are capable of removing large quantities of N from the soil. Results of field investigations on the application of dairy effluent to year-round forage crops have been reported by Woodard et al. (2002), Hubbard et al. (1987), Vellidis et al. (1993), and Newton et al. (1995). The general findings were that the amount of N removal by the crop and the NO_3^- -N in the soil water below the root zone tended to increase with increasing loading rates of N.

Nitrogen is present in the liquid manure in organic N (ON) and NH_4^+ forms. The latter is immediately available for crops but the ON must be mineralized before it is available for plant uptake. ON and NH_4^+ are not very mobile in soil, however, NH_4^+ can be nitrified to NO_3^- in days to weeks which is freely transported through the soil by flowing water. Proper management of liquid manure to maximize the fertilizer value and minimize water quality degradation requires knowledge of the complex dynamic interactions described above.

Dairies may employ different strategies in applying the liquid manures on cropland that entail different N inputs and timing of the applications. When different approaches of manure applications are adopted, it is difficult to project the outcomes in terms of crop yields and nitrate leaching due to the dynamic and interactive processes involving the reactions of applied N, irrigation, and plant growth. The temporal accounting of these coupled N reactions can be accomplished by utilizing a computer model such as the ENVIRO-GRO model (Pang and Letey, 1998). The model allows the simulation of various dairy liquid waste management options on water and nitrate distribution in the soil profile as a function of time, the amount of deep percolation, the amount of leached nitrate, and crop yield relative to that of a non-stressed crop. The main features of the model are as follows: The one dimensional Richards equation, which describes transient water flow through soil, is combined with a plant water uptake function. The water uptake function is based on the potential evapotranspiration (T_p) and the matric and

osmotic head potentials of the soil water. The convection-dispersion equation is used to describe chemical flow. The model allows additional water and/or N uptake from zones in the root system where they are adequate to compensate for deficiency in other sections of the root zone. Since potential water and N uptakes are related to plant growth, a feedback mechanism is programmed so that reduced growth results in reduced potential water and N uptakes.

The goal of the research reported here was to use the ENVIRO-GRO model to simulate how the amount of applied N, timing of N applications, ON mineralization rates, chemical form of N applied, and irrigation uniformity affected (1) yields of corn (*Zea Mays*) in summer and a forage grass in the winter in a Mediterranean climate and (2) the amount of NO_3^- leached below the root zone. The results can be used to guide the selection of management options to achieve desired goal.

E.2 Simulated Farm Management System

The cropping system typically used by dairy farmers in the San Joaquin Valley of California consists of planting silage corn in the spring and harvesting it in the fall, followed by a forage crop that is planted in November and harvested in April. In the simulations, we matched the irrigation and N applications with the requirements for crop growth. Dairy lagoon water was used as the only N source for the crops. Simulated irrigation was applied every 15 days with a mixture of lagoon water and regular waters. The irrigation was based on the T_p of the preceding 15 days and the amount of lagoon water (i.e. N application) was based on the total potential N uptake (N_p) for a nonstressed crop during the succeeding 15 days.

The fractions of ON and NH_4^+ in lagoon water can be variable, but we chose equal concentrations of each which is about the average case in the San Joaquin Valley. Simulations were also conducted using only ON to more clearly identify the effect of mineralization on the results. The applied N was assumed to be uniformly retained in the top 20 cm of soil at the time of application and that NH_4^+ would have been nitrified to NO_3^- prior to the next irrigation when it could be transported by water.

E.2.1 Factors considered in the model

1. Organic nitrogen mineralization

Mineralization of N can be described using the first order decay equation

$$N_{\min} = A_o [1 - \exp(-\lambda t)] \quad (1)$$

where N_{\min} is the amount of mineralized N, A_o is the total amount of N in the organic material, t is time, and λ is the N mineralization coefficient.

Nitrogen mineralization is dependent on temperature (Frederick 1956; Campbell et al. 1971). Stanford et al. (1973) estimated the rate constant at different temperatures. The relationship between mineralization rate and temperature is commonly described as a Q_{10} for a two-fold increase in the rate constant occurs for each 10° C rise in temperature.

The large concentrated animal feeding operation wastes are applied on land year round and a given field may receive multiple applications in a year. Tracking of mineralized N over a long-term becomes problematic when ON is applied multiple times and the temperature changes seasonally. We developed a computation algorithm to account for mineralized N over time resulting from multiple ON applications and temperature that changes seasonally.

The first order decay described by equation 1 was selected for ON mineralization. A standardized reference time t_0 must be selected as the reference point for counting. For convenience we chose January 1 as t_0 . Inputs to the model which are supplied by the user are the times and amounts of ON applications and the values of λ for various time periods of the year based on seasonal temperature. The time of applications are specified relative to t_0 . In multi-year simulations, the time counting in subsequent years are specified relative to the initial reference time. In other words, January 1 of the second year would be specified as day 366.

The algorithm keeps track of the N mineralization of each ON application and its seasonal changes of λ according to the input data. For each ON application the A_o equals the total amount of ON of this application and t in the program is set as 0 for the N mineralization computation. However the time for tracking application in mineralized N corresponds to the standard reference time.

When λ changes in the course of time, t in the program is reset to 0 and A_o is reset to be the total amount of remaining ON from the original application. The computation continues until λ is changed again. This way each application of ON has its own mineralization series which is tracked with respect to time. The total mineralized N at a given time is the sum of mineralized N from each prior application.

Information on mineralization rate coefficients of ON in lagoon water is generally lacking. Van Kessel and Reeves (2002) determined the mineralization rate of 107 dairy manures collected in five states in the Eastern United States. The manures were mixed with soil and incubated at 25° C for 56 days to determine mineralization rate. The manures had highly variable N mineralization characteristics including 13 samples that had net immobilization. The mean mineralization from all samples had a 280-day half-life. Nine samples had a 90-day or less half-life. These data clearly established the fact that mineralization rates are highly variable and very difficult to establish for a specific situation. We chose the 90- and 280-day half-lives for our simulations to determine the effects of mineralization rates on the results. The value of λ in equation 1 equals 0.0025 d⁻¹ for the 280-day half life and is equal to 0.0077d⁻¹ for the 90-day half-life.

Nitrogen mineralization rate varies with temperature. The λ value stated above was used for the months of May through October which are the warmest months. The value of λ for March, April and November, which have the intermediate temperatures, was set at 1/2 of the summer mineralization rate; and λ for December, January, and February, which are the coldest months, was set at 1/4th of the summer mineralization rate. More detailed refinements are probably not necessary based on the overall uncertainty of mineralization rates.

2. Plant nitrogen uptake

The potential N uptake rate (N_p) as a function of time is required input data. The total N uptake for a well-fertilized corn crop was measured as a function of time in the San Joaquin Valley for three years. The total uptake varied slightly between years, so the data were standardized by setting the maximum uptake to 1 for each year. The standardized N uptake was plotted as a function of time. The sigmoid relationship between standardized N uptake (N_s) and time was found to fit the equation

$$N_s = a + b/[1 + \exp(-(t - c)/d)] \quad (2)$$

where the coefficients were $a = 0.018$, $b = 0.99$, $c = 1273$, and $d = 232$, and the r^2 of the regression equation between computed and measured N_s was equal to 0.99. The derivative of that curve was used to compute the standardized potential N uptake rate as a function of time. These values were multiplied by 300 to calculate the actual potential N uptake rate as a function of time when the total potential N uptake was 300 kg ha⁻¹.

The cumulative N uptake for several forage varieties were measured as a function time in the San Joaquin Valley. The winter forage N uptake varied among the four varieties evaluated. We selected the Triticale (T2700) for our simulations. The functional relationship between N_s versus time for this forage was identical to equation 2. The data fit this relationship with r^2 of 0.99 when $a = 0.015$, $b = 0.97$, $c = 2038$, and $d = 348$. The maximum N uptake for this forage was 300 kg ha⁻¹. The potential N uptake rate is depicted as a function of time for the corn and forage crops in Fig. 1.

The model computes the nitrogen uptake relative to that of a nonstressed crop (RN_{up}), thus a relationship between relative yield (RY) and RN_{up} is required to convert the results into yield. Pang and Letey (1998) found that the relationship between N uptake and yield reported by Sexton (1993) for corn grown in Minnesota was almost identical to the results measured in California by Broadbent and Carlton (1979). The relationship we used was $RY = 1.7 RN_{up} - 0.70 RN_{up}^2$. The data for the forage used in our simulations was $RY = 2.03 RN_{up} - 1.03 RN_{up}^2$ based on the field data reported above.

3. Limiting nitrogen concentration

A relationship between the concentration of NO_3^- in soil solution (C_N) and a crop N stress factor (γ) must be established. The following rationale was used to establish γ values. On a field basis, transpiration rate has units of $m^3 m^{-2} d^{-1}$ and N uptake rate has units of $kg m^{-2} d^{-1}$. Nitrogen uptake rate divided by transpiration rate has units of $kg m^{-3}$, which are units for concentration. When $C_N \geq N_p / T_p$, γ was assigned a value of 1.0 (no stress). The concentration of N in the water carried to the root by the transpiration stream was adequate to meet potential N demand ($C_N T_p = N_p$). When $C_N \leq N_p / T_p$ (or $C_N T_p \leq N_p$), γ was assigned the value of $C_N / (N_p / T_p)$. The critical value of C_N (C_N^*) below which N uptake will be limiting is defined as N_p / T_p .

The agreement between simulated and measured experimental corn N uptake during the summer reported by Pang and Letey (1998) provides evidence that this relationship is appropriate for corn. However during the winter, T_p can be very low compared to N_p which results in a very high N_p / T_p ratio and thus an excessively high calculated C_N^* . Under these conditions a value of C_N^* must be selected based on experimental information. The value of $5 mg L^{-1}$ was selected based on the measured NO_3^- concentrations in the soil water below the root zone reported by Woodard et al. (2002) from studies applying dairy effluent to forage systems in Florida.

E.3 Irrigation Uniformity

The simulated results are for the condition that the irrigation and nitrogen applications were uniform across the field; however, this condition is rare in an agricultural operation. The approach proposed by Letey et al. (1984) was used to determine the impact of nonuniform irrigation on the results. Because nitrogen was applied with the water in our case, zones receiving more water also received more nitrogen.

For any point "a" (finite size but small enough to be considered uniform), infiltrated water (IW) can be related to the average water application on a field basis (AW) by

$$IW(a) = \beta(a)AW \quad (3)$$

where $\beta(a)$ is a parameter whose distribution over the field must be determined. For computational convenience, the distribution function of β can be approximated by a discrete

distribution in which β takes on only a finite number of distinct values which have known probabilities. Two arbitrary IW distributions in addition to the uniform case were chosen for analysis. For clarity in reporting the results the Christensen's uniformity coefficient (CUC), commonly used by irrigation engineers to express the degree of uniformity, was calculated for each distribution. CUC equals $100 [1 - (\sum x) / Mn]$ where x is the absolute value of the deviation from the mean, M , of the individual observations and n equals the number of observations. Two distributions which were symmetrical around the mean with CUC equal to 73 and 86 were used in the analysis. The simulation was conducted for a given amount of water and N application to each discrete fraction of the field and then the results were integrated for the entire field.

E.4 Simulated Variables

Field-average water application equal to $1.15 T_p$ for the 15-day period since the last irrigation was used in all simulations. During the period between crops, it was assumed that there was no evaporation from the field. The potential water loss between the time of the last irrigation and the harvest of the crop was applied as an irrigation at the beginning of the next crop season.

The several variables combinations which were simulated are summarized in Table 1 for uniform irrigation and Table 2 for nonuniform irrigation. The amounts of applied N were 1.0, 1.2, and 1.4 times the N uptake for a non-stressed crop. When the N was applied with each irrigation, the amount of N applied at each irrigation was related to the potential N uptake for the succeeding period of time until the next irrigation. In some cases the N was applied with one-third at time of planting and then one-third each at 30 and 75 days after planting. The N applied in all of the above stated simulations were equally divided between ON and NH_4^+ .

Other simulations were conducted when the applied N was entirely ON. These simulations had the total N being applied at the time of planting, and also having the N equally applied at 0, 30, and 75 days after planting. All simulations were conducted with the summer time mineralization rate one-half lives of 90 and 280 days.

The effects of irrigation uniformity were investigated by doing simulations with irrigation uniformity CUC values of 73, 86, and 100. The N sources were equally divided between ON and NH_4^+ and applications were with each irrigation and also at three times during the cropping season.

E.5 Input Data for the Model

The simulations were conducted for a soil bulk density of 1.40 g cm^{-3} and a saturated water content of $0.48 \text{ cm}^3 \text{ cm}^{-3}$. The saturated hydraulic conductivity was chosen at 2.0 cm hr^{-1} . The parameters used in the Hutson and Cass (1987) hydraulic function were as follows: water content at the inflection point (θ_i) was $0.48 \text{ cm}^3 \text{ cm}^{-3}$; the matric potential at the inflection point (h_i) was -0.0028 MPa ; the air entry matric potential (a) was -0.0027 MPa ; and exponent (b) of the equation relating matric potential to water content was 3.8. The exponent (b_{hb}) for the equation

relating hydraulic conductivity to water content was set as 15.0. These functions are typical for a loam soil.

The lower boundary was set at 2m with 5-cm increments of soil depth for computation. The bottom boundary condition was set as free drainage. The upper boundary condition was set as flux control conditions with infiltration of irrigation according to the input rate. The bottom of the root zone was set at 1.5 m where drainage and N leaching were calculated.

The values, in units of MPa, for the threshold matric water stress (h_t) was equal to -0.05 for both crops and the matric stress causing 50% growth reduction (h_{50}) was equal to -0.14 for corn and -0.26 for forage. Irrigation water, even with lagoon water mixed in, was assumed to be sufficiently low in salinity to not affect plant growth.

The initial water content distribution was established by setting the soil profile at saturation and then allowing redistribution for 14 days with free drainage as the bottom boundary condition. This resulted in a water content of $0.34 \text{ cm}^3 \text{ cm}^{-3}$ and the matric potential equal to -0.012 MPa at the bottom boundary and $0.32 \text{ cm}^3 \text{ cm}^{-3}$ at the upper boundary. This soil water content profile was taken as the initial water content condition for corn in the first year, thereafter continuous simulation was conducted. The initial inorganic N distribution was 150 kg ha^{-1} evenly distributed over the top 20 cm and 100 kg ha^{-1} evenly distributed over the 20-200 cm layer. The initial water and N distribution only affected the results for the first and sometimes second year of the multiyear simulations in a manner similar to how the initial soil condition affects results in the field. The reason for running multiyear simulations was to determine the long term consequences of a management scheme and eliminate the effects of the initial conditions.

The T_p was taken as the reference ET_0 times a crop coefficient. ET_0 values for Fresno, California, and crop coefficients for corn were taken from a report by Letey and Vaux (1985). The crop coefficient for the forage crop was assumed to be 1.0.

E.6 Results

The organic N mineralization rate is plotted as a function of day of year for five years in figure 2 when $N_p = 1.4$, summer half-life is 280 days, and the N was applied with each irrigation. Note that steady state values are reached after about five years. The temperol rate of mineralization does not coincide with the temperol rate of N uptake (Fig. 1). Therefore the timing of leaching events will significantly affect the results. Large leaching rates at the initial and final stages of corn growth would particularly cause much N leaching that could affect crop yields as well as ground water degradation. In our simulations no large leaching events were programmed.

The relative yield (RY) of corn and forage and the amount of NO_3^- -N leaching (NL) are shown in Figure 3 over a 10-year period. The simulated conditions in this case were uniform application of $1.2 N_p$ with every irrigation of $1.15 T_p$. The applied N was equally divided with 50 % NH_4^+

and 50% ON, where the ON had 280- or 90- day summer-time half-life mineralization rate. Maximum yields for both crops were simulated throughout. The high yields during the first one or two years can partially be attributed to the programmed high initial inorganic nitrogen content in the profile at the beginning of the simulations.

Note that the NL with a few exceptions increases with time and approaches a steady state value which is reached earlier for the 90-day than for the 280-day half-life. Eventually both reached approximately the same value. One of the important findings of these analyses is that when a given amount of organic N is applied consistently, the annual amount of mineralized N eventually equals the amount of total applied N regardless of the mineralization rate constant. This is a fortunate circumstance because accurate information on mineralization rates is usually lacking. The mineralization rate constant determines the time period required to achieve steady state condition, but not the eventual quantitative steady state rate.

Note that after the first year essentially no NL occurs from the forage crop. (The first year results are greatly affected by the initial soil N distribution that was selected.) The low leaching under the forage crop can be attributed to two factors. The crop is grown in the winter months when the mineralization rate is very low. Therefore very little of the ON applied to the forage crop is mineralized and available for either crop uptake or leaching. The forage crop therefore was partially dependent on the mineral N remaining after the corn crop. The forage crop therefore utilized the available inorganic N sources which left little for leaching.

Much ON is mineralized between the period of forage production and maximum n uptake by the corn crop as well as after the time of maximum N uptake. These factors contribute to the significant amounts of computed NL under the corn crop.

Programming N application with time and amount consistent with crop uptake is simple when conducting computer simulations. Under a farm operation it is more common that the dairy waste only be added during some irrigations. For comparative purposes, we simulated applying the lagoon water during three irrigations for each crop, with equal amounts applied at the beginning, and approximately one third and two thirds through the crop season. All other conditions are the same as reported for Figure 3. The results of the latter simulation are illustrated in Figure 4. Note that in this case, corn yields were decreased and because of the reduced corn yields, more NL occurred. These results identify the importance of the timing as well as the total amount of N application. The decrease in corn yield is a result of having inadequate mineral N available during the relatively short time of peak N uptake requirement by the corn crop (fig. 1). Much of the applied mineral N and also the amount of mineralized ON occurred after the peak crop requirement. Note that the forage was not affected because it benefited from the carry over from the corn crop. These results illustrate one of the challenges of managing organic N such that the availability of mineral N matches the crop N requirement on a temporal basis.

The temporal effects reported above are largely associated with the mineralization of organic N. To more completely understand the dynamics of N mineralization on crop yield and nitrate leaching, simulations were conducted for the same conditions as depicted in Figure 4 except all of the N was applied in the organic form. Results of this simulation are illustrated in Figure 5. Note, in comparing the results depicted in figures 4 and 5 that the main difference between the presence or absence of mineral N occurs during the first few years of the simulation. After the organic N has been applied for sufficient years to achieve steady state, the response was very similar to a combination of organic and mineral N as long as the total amount of N applied was identical.

Simulations where higher and lower N applications than those depicted in figures 3, 4, and 5 were conducted. The main findings were as expected that higher N applications generally lend the higher yields and NL and the opposite for lower N applications. The temporal effects were not sensitive to total N application.

The results presented thus far are for uniform irrigation. The effect of the irrigation uniformity on results for the fifth year of simulation when steady state conditions had been approached are reported in Table 3. The simulated results are for N application with every water application and also for N application three times during each cropping season. The average water application was 1.15 T_p .

In general, RY increases with increasing irrigation uniformity. However, the difference between a CUC equal to 86 and uniform irrigation is not great. Increasing the N application rate tended to increase RY and NL.

The effect of irrigation uniformity on the amount of NL is variable. Indeed, when the N is applied three times during the cropping season, increasing irrigation uniformity resulted in a simulated increase in the amount of nitrate leached. Non-uniform irrigation results in parts of the field being "under irrigated" and other parts of the field being "over irrigated". Since the nitrogen was applied with the water, the sections of the field receiving the least amount of water also received the least amount of nitrogen. However, water rather than nitrogen was the limiting growth factor and no deep percolation of water occurred on the drier parts of the field. This would allow a small fraction of applied nitrogen to accumulate in the drier parts of the field thus leading to an overall field average reduced nitrate leaching.

Application of only organic N once at the beginning of each crop was compared to application three times during each crop growing season. The general results are as follows. During the first year, the yields for both corn and forage were higher for the one time application but in succeeding years, there was no difference between the two options. Under the steady state condition, application of 1.2 N_p resulted in a relative corn yield of 0.9 and maximum yield for forages. Increasing the nitrogen application to 1.4 N_p increased the relative corn yield to .97.

However, increasing the N application to 1.4 N_p also increased the N leached by about 85 kg ha⁻¹ yr⁻¹.

E.7 Conclusions

One major conclusion from this study is that when applying ON ultimately steady state conditions are achieved, whereby an equivalent amount of nitrogen applied in the organic form will be mineralized during a year. Steady state conditions are achieved more rapidly for materials with higher mineralization rates. This finding also underlines the importance that the results from short-term field experiments must be interpreted with caution. The experimental results will be very dependent upon the initial N status of the soil, mineralization rate of applied material, and whether organic N had been applied to the field several years prior to the experiment. When transitioning to an ON fertilizer source, higher amounts should be applied during the initial years and then decreasing amounts in successive years as steady state mineralization is approached.

A second conclusion is that the timing and total quantity of N application are both very important in affecting crop yield and potential N leaching. Many crops have very high N requirement over a relatively short period of time and will experience reduced growth if adequate N is not available during that period. Because mineralization of N is a continuous function, the timing of N availability with crop requirement is difficult to synchronize (Pang and Letey, 2000). Significantly higher simulated yields were achieved when N was applied with every irrigation to meet crop demands as compared to equal applications three times during the crop season (figures 3 and 4).

Increasing irrigation uniformity resulted in increasing yield for a given N application amount. Increasing irrigation uniformity increased, decreased, or had almost no effect on the amount of N leaching depending on the specific scenario. Because the N was applied with the water, nonuniform irrigation also caused nonuniform N application which contributed to the variable effects on N leaching.

Planting a forage crop during the winter effectively reduced the leaching of residual soil N following the corn crop. Application of ON during the winter when mineralization is slow provides very little mineral N for winter crop, but it becomes a major N source for the summer crop.

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Table 1.

Summary of the combinations of variables which were simulated under uniform irrigation.

N amount	N composition	N appl. timing ^b	Mineralization half-life days
1.0 N _p ^a	NH ₄ ⁺ and ON	Each irrig.	90 and 280
1.2 N _p	NH ₄ ⁺ and ON	Each irrig.	90 and 280
1.4 N _p	NH ₄ ⁺ and ON	Each irrig.	90 and 280
1.0 N _p	ON	0, 30 and 75	90 and 280
1.2 N _p	ON	0, 30 and 75	90 and 280
1.4 N _p	ON	0, 30 and 75	90 and 280
1.0 N _p	ON	0	90 and 280
1.2 N _p	ON	0	90 and 280
1.4 N _p	ON	0	90 and 280

^a N_p is annual potential N uptake by crop^b The numbers indicate days after planting

Table 2.

Summary of the combinations of variables which were simulated under nonuniform irrigation.

N amount	N composition	N appl. timing ^b	Mineralization half-life days	CUC ^c values
1.0 N _p ^a	NH ₄ ⁺ and ON	Each irrig.	90 and 280	73, 86, 100
1.2 N _p	NH ₄ ⁺ and ON	Each irrig.	90 and 280	73, 86, 100
1.4 N _p	NH ₄ ⁺ and ON	Each irrig.	90 and 280	73, 86, 100
1.0 N _p	NH ₄ ⁺ and ON	0, 30, 75	90 and 280	73, 86, 100
1.2 N _p	NH ₄ ⁺ and ON	0, 30, 75	90 and 280	73, 86, 100
1.4 N _p	NH ₄ ⁺ and ON	0, 30, 75	90 and 280	73, 86, 100

^a N_p is annual potential N uptake by crop^b The numbers indicate days after planting^c Christensens uniformity coefficient

Table 3. Simulated effects of irrigation uniformity on yield and nitrate leached.

Application	N applied with every irrigation											
	CUC = 73				CUC = 86				CUC = 100			
	RY _c ^a	NL _c ^b	RY _f ^c	NL _f ^d	RY _c ^a	NL _c ^b	RY _f ^c	NL _f ^d	RY _c ^a	NL _c ^b	RY _f ^c	NL _f ^d
1.0 N _p	0.85	64	0.92	15	0.91	37	0.94	3	0.95	0	0.97	0
1.2 N _p	0.92	127	0.97	42	0.98	111	1	23	1	114	1	0
1.4 N _p	0.92	178	0.97	76	0.98	163	1	58	1	182	1	48

Application	N applied 0, 30, and 75 days after planting											
	CUC = 73				CUC = 86				CUC = 100			
	RY _c	NL _c	RY _f	NL _f	RY _c	NL _c	RY _f	NL _f	RY _c	NL _c	RY _f	NL _f
1.2 N _p	.77	134	.96	6	.83	146	1	8	.84	202	1	3
1.4 N _p	.83	158	.97	34	.91	162	1	39	.93	232	1	51

^a Relative yield of corn

^b Relative yield of forage

^c Amount of N leached under corn (kg ha⁻¹)

^d Amount of N leached under forage (kg ha⁻¹)

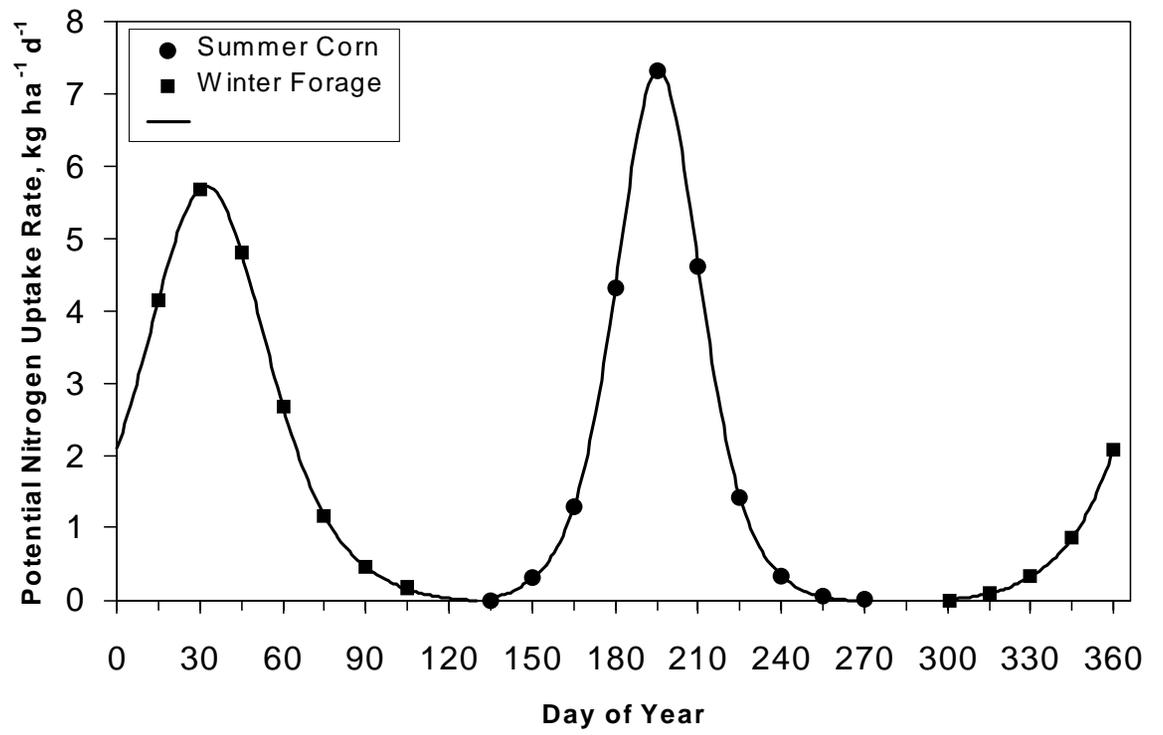


Figure 1. The potential N uptake rate as a function of time for corn and forage crops.

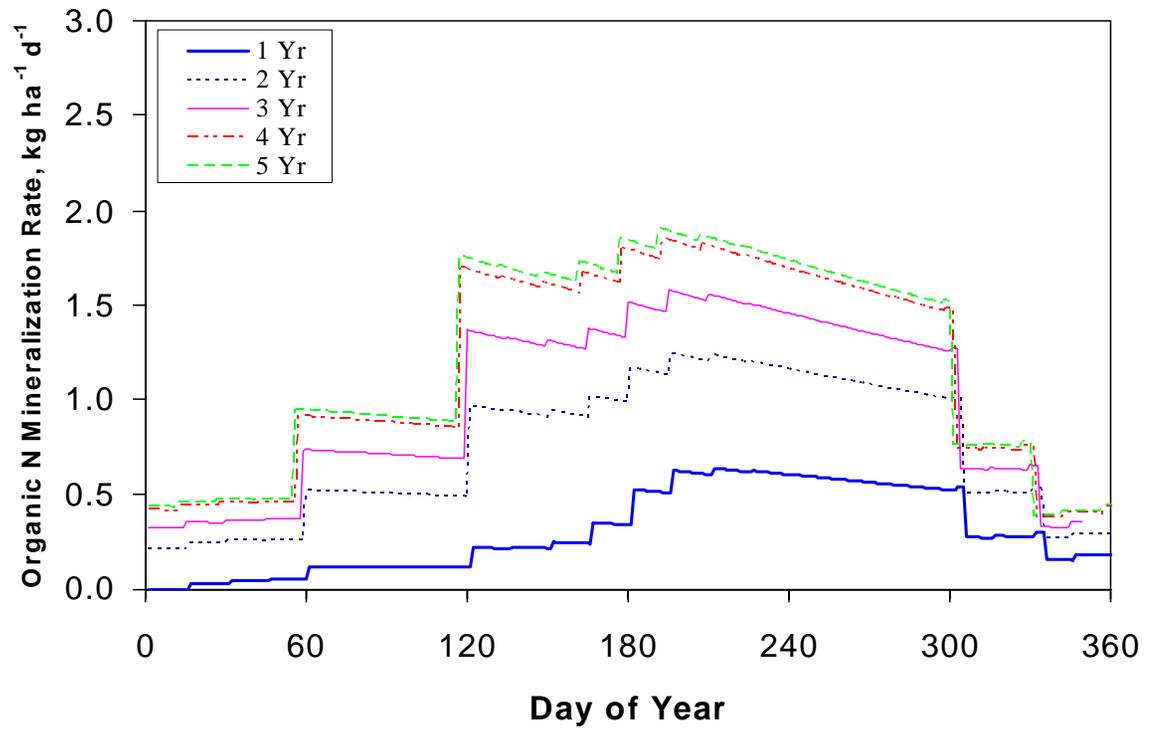


Figure 2. The organic N mineralization rate as a function of time for five years when the time and amount of organic N application was biweekly to meet 70 percent of the potential N uptake and the half-life was 280 days during the summer. The results for years greater than 5 are identical to the fifth year results.

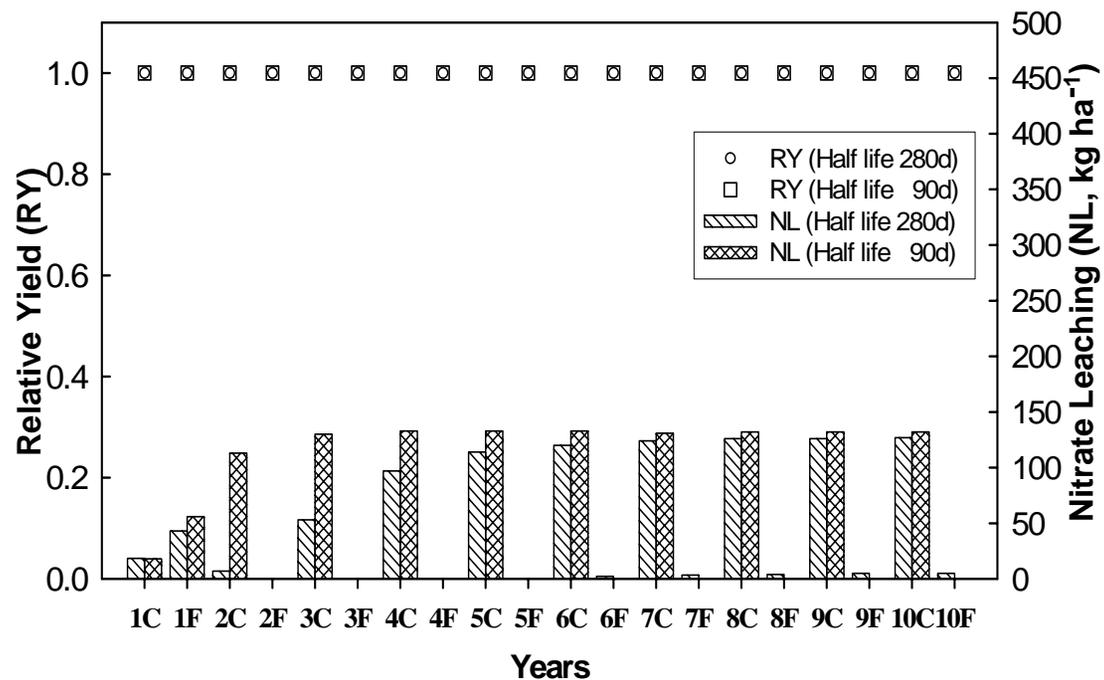


Figure 3. The relative yield of corn (C), and forage (F) and the amount of nitrate-nitrogen leached during individual crop seasons as depicted over a ten-year period under uniform irrigation. The lagoon water was applied each irrigation with the total N equal to 1.2 times the potential N uptake. The results are depicted for summer time half life mineralization rates of 90 and 280 days.

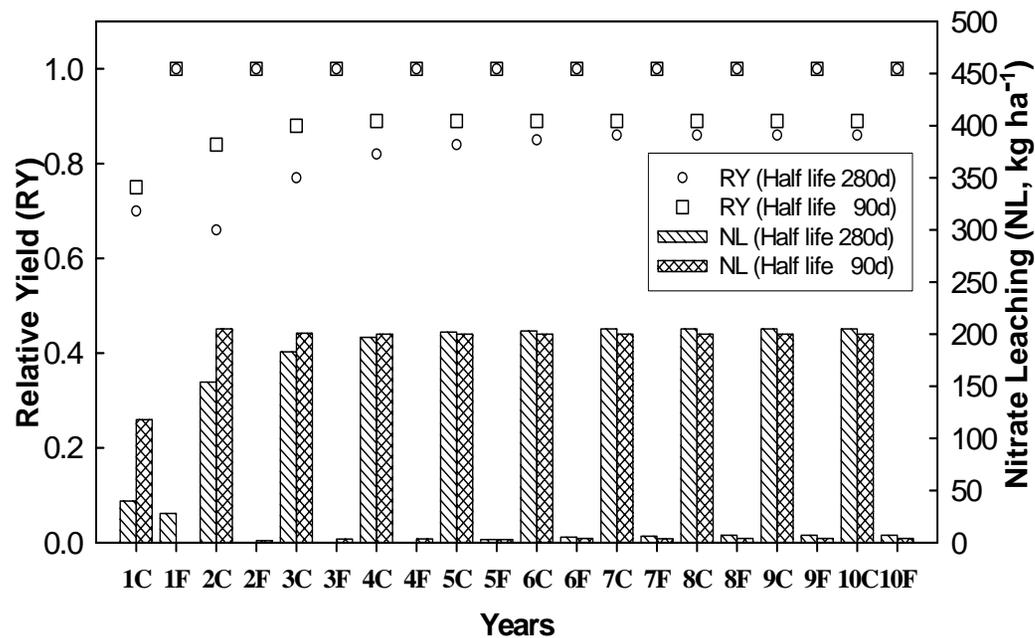


Figure 4. The relative yield of corn (C), and forage (F) and the amount of nitrate-nitrogen leached during individual crop seasons as depicted over a ten-year period under uniform irrigation. The lagoon water was applied three times during the growing season with the total N equal to 1.2 times the potential N uptake. The results are depicted for summer time half life mineralization rates of 90 and 280 days.

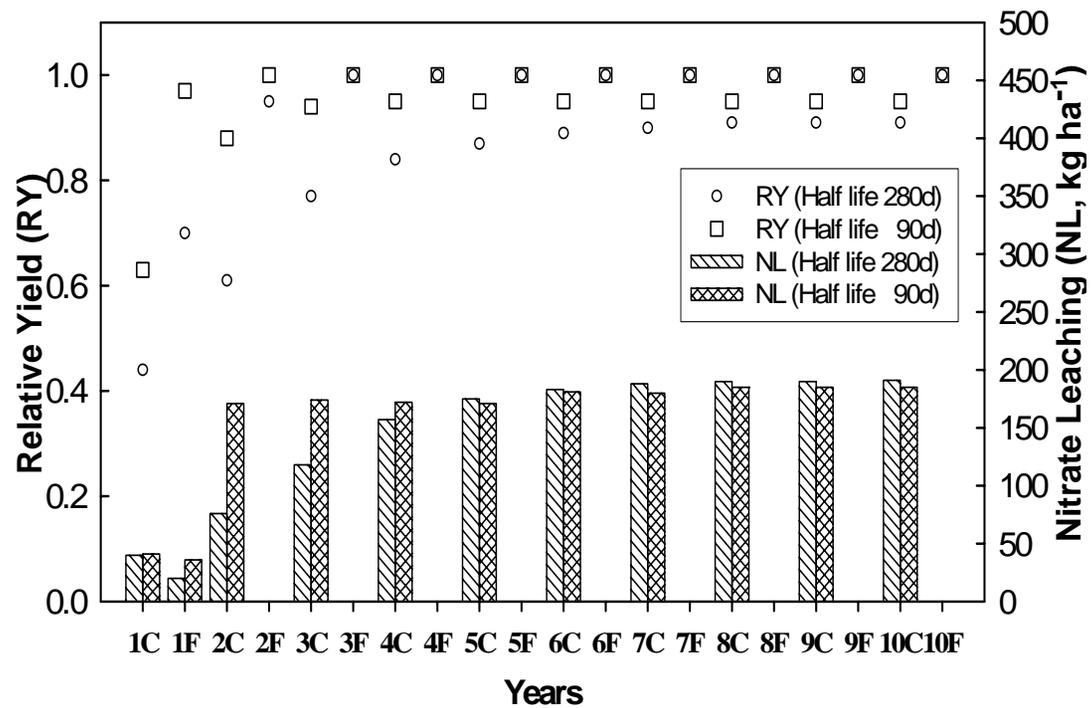


Figure 5. The relative yield of corn (C), and forage (F) and the amount of nitrate-nitrogen leached during individual crop seasons as depicted over a ten-year period under uniform irrigation. Only organic N was applied three times during the growing season with the total N equal to 1.2 times the potential N uptake. The results are depicted for summer time half life mineralization rates of 90 and 280 days.

APPENDIX F:

Field Studies of N Use Efficiency

Andrew Chang and Stu Pettygrove

N Use Efficiency can be expressed in various forms, but in general refers to crop recovery (either as plant uptake or harvested product) of N inputs to the crop-soil system. Many studies of crop N recovery have been published in the scientific literature. Most encompass only one season; few studies of multi-year recovery have been conducted.

Lund et al. (1978) conducted a 12-year investigation on the N utilization efficiency in commercial vegetable production fields in Santa Maria Valley of California. They developed an N balance showing that 30% of the applied N was removed in the harvested crops, 37% was leached below the root zone, and 33% was unaccounted for, which was attributed mainly to gaseous loss of N due to denitrification. With a crop harvest N removal of 30%, an application of 3.33 times (333%) times harvest N removal would be required. If nitrate leaching were eliminated, the N input could be reduced by 37% (i.e., the leached N) and the applied N requirement from all sources would be 1.49 times (149%) of harvest N removal.

Broadbent and Carlton (1979) conducted field experiments on fertilizer N use efficiency under irrigated cropping conditions in Davis, CA and the UC Kearney Agriculture Center near Fresno, CA, using ¹⁵N-labelled fertilizer. They reported that applied N recovery ranged from 35 to 68% at the Fresno site and from 30 to 67% at Davis, with maximum recovery occurring at application rates that produced maximum grain yield. Very little nitrate derived from fertilizer escaped from the root zone unless fertilizer rates were in excess of crop needs. They concluded that “the potential for excess nitrate in the profile rises sharply above the optimum N fertilization rate.” They also concluded that “optimum production of corn is compatible with minimum pollution hazard with careful management of fertilizer and irrigation.” Under the circumstances, the N input requirements from all sources corresponding to the agronomic rates would be those with the highest fertilizer N recovery rates, calculated to be 148%.

Bock and Hergert (1991) provided general guidelines for unit N fertilization (e.g., N input per bushel harvested grain) based on assumed fertilizer N recovery by the above-ground crop. They recommended that the fractional fertilizer N recovery by upland cereal grains should be set at 0.45 when timing of applications was poor, 0.60 for medium efficiency, and 0.70 in highly efficient situations. Their definition of efficiency referred only to commercial N fertilizers applied in the year of crop production and did not include N derived from mineralization of soil organic matter, crop residues, or organic amendments like manure. From the minimum N rates required for maximum yield, Bock and Hergert (1991) derived the criteria in Stanford (1973) that 50 to 70% recovery of applied N is physically possible in most soil-plant systems, and up to 30 to 50% of applied N can be immobilized during decomposition of corn stover and roots if that much inorganic N remains in the root zone at the end of the growing season (Bock, 1984).

Based on Bock and Hergert (1991) and Stanford (1973) the agronomic rate again is approximately 140% of the crop N requirement.

Raun and Johnson (1999) estimated that worldwide, N use efficiency for cereal production (rice, wheat, corn, barley, millet, oats, and rye) is approximately 33%. They compared estimates of worldwide fertilizer N use on cereal grains and estimates of N removed in harvested grain. They did not include as inputs soil N obtained from biological N fixation (e.g., when cereals are grown in rotation with legumes) or miscellaneous other sources such as direct deposition of atmospheric N and nitrate contained in irrigation water. Based on their analysis, it is possible only to state that the long-term N removed worldwide by harvests of cereals amounts to between one-third and two-thirds of all externally derived N inputs. Based on this set of data, the N input requirement (from all sources) for cereal production, based on the average N recovery percentage, is 150 to 300% of harvest removal.

A few N recovery studies have been conducted in manured systems. It is much more difficult to label manure N with the tracer ^{15}N than to do so with synthetic inorganic fertilizers. In a three-year study with potted plants using ^{15}N -labelled poultry manure, the N recovery by cereals ranged from 19 to 36% (fresh manure), 17 to 24% (anaerobically incubated) or 12 to 14% (aerobically incubated). On average, 62% of manure N was found in the soil after three years. Gaseous losses ranged from 7 to 26% of N (Kirchmann, 1989). It appeared that the recovery of organic-borne N by plants was consistently lower than from chemical fertilizers and the remainder of the added N accumulated in the soils.

Powlson et al. (1991) conducted ^{15}N -labelled fertilizer trials in Rothamsted Station, UK and reported that 50 to 80% of fertilizer N was found in the harvested crop and 10 to 25% in soil. It was noted that most of the labeled N was in organic forms, not as “left-over” fertilizer N. Long-term studies at Rothamsted Station, UK, on a silty clay loam and a sandy loam, showed that more than 89 lbs N acre⁻¹ yr⁻¹ (100 kg N ha⁻¹ yr⁻¹) were lost from soils when large applications of animal manure, sewage sludge and composts were being applied (Addiscott et al., 1991). The mineralization of organic N over time increases soil N and correspondingly decreases the nitrogen input requirement.

The Sustainable Agriculture Farming Systems (SAFS) project at UC Davis provided multi-year crop N recovery estimates for irrigated cropping systems employing animal manure applications. The crop rotations include processing tomato, corn for grain, wheat for grain, and green manure legume crops. An organic farming system was compared to two-year and four-year conventional rotations and a low-input system. The organic system relied on manure and legume cover cropping for N supply. Poudel et al. (2001) evaluated the N balance for the first 10 years of the SAFS project. N inputs and crop harvest removal of N were higher in the conventional systems than in the organic system. The apparent N recovery was 48% in the organic system, 71% in the conventional two-year rotation, and 73% in the conventional four-year rotation. Thus, in the organic system, more than 2 lbs N were added from all sources for each pound of N removed in the harvested crop, compared to an input of 1.4 lbs N for each pound harvested in the

conventional systems. However, the apparent N recovery was not related to the amount of N leached in each treatment. The annual loss of N averaged over 10 years was 8 lbs N acre⁻¹ yr⁻¹ (9 kg ha⁻¹ yr⁻¹) in the organic system, while in the conventional four-year rotation treatment it was 37 lbs N acre⁻¹ yr⁻¹. In the organic system, the buildup of soil organic N averaged 80 lbs N acre⁻¹ yr⁻¹ (90 kg ha⁻¹ yr⁻¹) much higher than the annual 7 lbs N acre⁻¹ yr⁻¹ (8 kg ha⁻¹ yr⁻¹) buildup in the conventional four-year rotation treatment. Higher soil organic N may have contributed to higher denitrification rates as well.

While the organic system appeared much less efficient than the conventional four-year rotation in terms of N recovery by crops (apparent N recovery of 48% vs. 73%), the nitrate leaching was much lower in the organic system due to the increase of soil organic matter and the accompanying build-up of soil organic N. However, the build-up cannot continue indefinitely. At some point, a steady state will be reached, and the net N mineralization of soil organic matter will increase to the point that annual N inputs can be reduced, thus increasing the apparent N recovery for that system. Note that the organic farming system in the SAFS study received considerably lower amounts of organic N than do dairy forage fields in the San Joaquin Valley (Harter et al., 2001). Also, the type of cropping system and the type of manure applied (poultry waste) are distinctly different from typical dairy manure systems.

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APPENDIX G:
Reality Check - Nitrogen Mineralization and Leaching Losses
Estimated from Shallow Groundwater Observations in a Dairy System
with Sandy Soils

Thomas Harter and Andrew Chang

Under conventional practices (no specific manure nutrient management), Harter et al. (2002) estimated that the nitrate N and dissolved salt loading from five dairies in the northeastern Central Valley (Merced and Stanislaus County) averaged 280 and 4,300 kg ha⁻¹ yr⁻¹, respectively. These estimates are based on direct measurements of shallow groundwater known to originate primarily from the dairies and their surrounding fields. Under the supervision of Thomas Harter, Van der Schans (2001) confirmed the leaching rates with a calibrated, site-specific groundwater model. Harter et al. (2001) and Van der Schans (2001) also implemented a field mass balance for a field in Merced County that received frequent manure applications and compared the leaching estimates from the mass balance with those obtained from groundwater quality measurements downgradient of the field. They estimated that, on the average, the N input to the forage production field amounted to approximately 1,200 kg ha⁻¹ yr⁻¹, of which 900, 280, and 28 kg ha⁻¹ yr⁻¹ came from dairy manure, chemical fertilizers, and irrigation water plus atmospheric deposition, respectively. A total of 540 kg ha⁻¹ yr⁻¹ of the 1,200 kg N ha⁻¹ yr⁻¹ input (45%) may be accounted for by N in the harvested corn and fall-planted wheat. The mass balance approach suggests that 660 kg ha⁻¹ yr⁻¹ are lost to leaching, volatilization, or denitrification. Monitoring well measurements on the other hand indicate that approximately 600 to 700 kg ha⁻¹ yr⁻¹ N was recharged to the shallow groundwater (55%). Hence, groundwater nitrate accounted for all of the losses, meaning that N was completely mineralized and no denitrification occurred. Does this mean that volatilization and denitrification losses are near zero? The authors point out that the estimate for the manure N input (900 kg ha⁻¹ yr⁻¹) is not very precise. Higher amounts may have been applied, which would imply that some denitrification and/or volatilization losses have indeed occurred.

Harter et al. (2001) installed a series of monitoring wells immediately downgradient from manure application fields to assess the condition of shallow groundwater and its response to improved nutrient management that began in 1998. By matching dairy wastewater nutrient application to crop nitrogen uptake during the growing seasons, they showed that nitrate concentrations in the groundwater have been steadily declining since the experiment began due to the improved balance between N input and crop N uptake. Comparison of mass balance analysis and groundwater quality also confirmed that in this case a) all organic nitrogen mineralized and b) that denitrification and volatilization during the application and in the unsaturated zone of the fine sandy to loamy sandy soils were negligible.

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APPENDIX H:

Fate of Liquid Dairy Manure Nitrogen in an Irrigated Double Crop Corn-Grain Rotation, California.

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Abstract

Manure nutrient management, long neglected and considered merely a waste disposal operation, is one of the key elements of sustainable dairy farming and the corner-stone of recently enacted local, state, and federal animal farming regulations in California. A four-year field trial of liquid dairy manure nutrient management in a flood irrigated corn-winter grain double crop rotation was implemented. Throughout the study period, water and nitrogen applications were metered, soil profile ammonia and nitrate measurements were taken, and shallow groundwater quality was measured immediately downgradient of the field trial. Here, we use inverse modeling of the unsaturated zone flow and transport processes to determine field scale soil hydraulic properties, mineralization, nitrification, and denitrification rates, and the linear sorption coefficients for organic nitrogen and ammonia. The retardation coefficient for organic nitrogen sorption is very high, although soil profile ammonia predictions are not very sensitive to retardation coefficients, R , above 10. The retardation coefficient for ammonia sorption ($R = 3$) is obtained by fitting modeled to measured plant nutrient uptake, which is very sensitive to ammonia sorption. The best fit transformation rates are $k_{\min}=1.6 \text{ d}^{-1}$ for summer-time mineralization, $k_{\text{nit}}=0.1 \text{ d}^{-1}$ for summer-time nitrification, and $k_{\text{den}}=0$. Despite the extensive soil profile data collected, results are relatively insensitive to mineralization rates. The model is insensitive to organic N transformation that results in half-lives much smaller than 2 months. Organic N half-lives ranging from less than 1 day to 2 months all give reasonable results. However, the inverse model is much more sensitive to ammonia nitrification rates. The ammonia half-life is found to be one week. Unlike soil ammonia and nitrate profiles, predicted nitrate fluxes to groundwater are sensitive to the percent organic N available for mineralization and net denitrification. Only under complete mineralization and with negligible denitrification can the model explain the elevated groundwater nitrate concentrations. The relatively high mineralization and nitrification rates are thought to be caused by the low organic carbon content in the soil profile, the high temperatures, the high dissolved oxygen content of the irrigation water, and the strongly oxic conditions throughout

the unsaturated zone. The same conditions also explain why denitrification is found to be minimal.

Keywords: nitrogen, mineralization, nitrification, sorption, vadose zone modeling, manure, dairy, fertilization

H.1 Introduction

Nutrient management is a key issue regarding the environmental sustainability of dairy operations. In California and elsewhere, manure nutrient management has become the corner stone of the permit process for dairy (and other animal) farming at the local, state, and federal level. Proper nutrient management is key to successfully growing a crop on manure fertilizer without negatively affecting the environment (groundwater, surface water, air). The goal of proper nutrient management is to balance the amount of nutrients applied in a field with the amount of nutrients used by the crop (Meisinger and Randall, 1991). Optimal nutrient application practices that meet crop needs while sustaining environmental quality rely on:

- knowing the amount of nutrients in the manure,
- knowing the amount and distribution of nutrients applied to the field,
- knowing the crop nutrient needs and uptake processes,
- and understanding the fate of nutrients once they are incorporated into field soil.

The focus of this report is the fate of soil nitrogen: In California, dairies are the largest animal farming industry. Most of California's dairy herd is located in valleys and basins, often far from surface water (e.g., Tulare basin in the Central Valley), but generally overlying alluvial aquifer systems that are more or less vulnerable to nonpoint source pollution. Of the major nutrients, nitrogen is of particular interest with respect to groundwater due to its potential for nitrate contamination. In contrast, the other two major nutrients (potassium and phosphorus) are not commonly found to occur at concentrations that would limit the beneficial uses of groundwater. Phosphorus becomes a major nutrient management issue only in areas with surface water runoff to streams or indirect runoff via tile drainage or via shallow groundwater to nearby streams. Potassium contributes to the overall salinity of groundwater.

Manure nitrogen is primarily available as organic nitrogen and ammonium-nitrogen (ammonium-N). In both, solid and liquid manure, nitrate-nitrogen (nitrate-N), nitrite-N, and other forms of inorganic nitrogen are generally available at negligible and often unmeasurable amounts. After the application of manure to field crops, organic nitrogen begins to mineralize to ammonium-N, which in turn converts to nitrate-N if soils are sufficiently aerated. It is generally thought, that only a fraction of the organic nitrogen will mineralize over time, while the remainder becomes incorporated into the soil nitrogen pool (bound into soil organic matter).

Knowledge of the amount and timing of organic nitrogen mineralization is critical in planning proper nutrient management. Mineralization rates not only affect the timing of nutrient applications. The total amount of mineralization is directly related to the sizing of the dairy herd given a limited acreage for manure fertilizer applications. The larger the amount of

organic N that becomes permanently embedded in the soil organic N pool, the larger the amount of manure that can be applied and the larger the herd size sustainable. In well-aerated soils, ammonia nitrification is usually complete and under California climatic conditions generally occurs within days of mineralization to ammonia. Only the nitrate-N is mobile enough to leach below the root zone, through the deeper vadose zone, and into shallow groundwater in significant quantities, where excess moisture leaches out of the soil profile.

Available literature for quantification of the nitrogen cycling processes in soils are climate-, soil-, and crop-specific. Little data are currently available for the typical cropping systems and soils occurring in the San Joaquin Valley, California, where two-thirds of the California dairy herd is located. Moreover, much of the existing literature relies on extrapolation of soil profile and plant uptake data with little or no control through deep vadose zone or shallow groundwater quality data. The objective of this report is to characterize mineralization, nitrification, and denitrification processes that occur in the vadose zone below a crop fertilized with liquid dairy manure. A four year field trial was established in a corn and winter grain rotation on a dairy farm located on relatively permeable, sandy soils near Hilmar, Merced County, California. The field, “Bun1”, was managed to optimize the use of liquid dairy manure, while minimizing the use of commercial fertilizer. We use inverse modeling of unsaturated zone flow and transport processes to estimate mineralization, nitrification, and denitrification rates from root zone nitrate-N and ammonium-N profiles collected at various time intervals and from groundwater nitrate measurements observed during the four year trial.

H.2 Model Description

The modeling period is from April 28th, 1998 to September 28th, 2001. We use HYDRUS-1D (ver.2) (Simunek et al., 1998).

H.2.1 Water transport in vadose zone

The governing water transport equation is expressed by Richards' equation:

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial z} \left[K(\theta) \frac{\partial h}{\partial z} + K(\theta) \right] \quad (1)$$

θ : volumetric water content (cm cm^{-3}), h : pressure head (cmH_2O), K : unsaturated hydraulic conductivity (cm h^{-1}), z : vertical coordinate (cm).

The relationship between h and θ is formulated by van Genuchten's equation:

$$\theta = \frac{\theta_s - \theta_r}{\left[1 + (-\alpha h)^n\right]^m} + \theta_r \quad (2)$$

θ_s : saturated volumetric water content, θ_r : residual volumetric water content, α , n , m : parameters of van Genuchten's equation ($m=1-1/n$)

The unsaturated hydraulic conductivity is also formulated by van Genuchten's equation.

$$K = K_s \sqrt{S_e} \left[1 - (1 - S_e^{1/m})^m\right]^2 \quad (3)$$

K_s : saturated hydraulic conductivity (cm h^{-1}), S_e : effective saturation

$$S_e = \frac{\theta - \theta_r}{\theta_s - \theta_r} \quad (4)$$

H.2.2 The fate of nitrogen

We consider organic-nitrogen (Org-N), ammonium-nitrogen ($\text{NH}_4\text{-N}$), and nitrate-nitrogen ($\text{NO}_3\text{-N}$). The transformations of nitrogen are simulated using HYDRUS-1D including mineralization (Org-N \rightarrow $\text{NH}_4\text{-N}$), nitrification ($\text{NH}_4\text{-N}$ \rightarrow $\text{NO}_3\text{-N}$), and denitrification ($\text{NO}_3\text{-N}$ \rightarrow gaseous nitrogen).

The governing equations describing the transport and fate of nitrogen are expressed as follows,

$$\frac{\partial \theta c_1}{\partial t} + \frac{\partial \rho s_1}{\partial t} = \frac{\partial}{\partial z} \left(\theta D_1 \frac{\partial c_1}{\partial z} \right) - \frac{\partial q c_1}{\partial z} - k_{\min} \theta c_1 - k_{\min} \rho s_1 \quad (5)$$

$$\frac{\partial \theta c_2}{\partial t} + \frac{\partial \rho s_2}{\partial t} = \frac{\partial}{\partial z} \left(\theta D_2 \frac{\partial c_2}{\partial z} \right) - \frac{\partial q c_2}{\partial z} + k_{\min} \theta c_1 + k_{\min} \rho s_1 - k_{\text{nit}} \theta c_2 - k_{\text{nit}} \rho s_2 \quad (6)$$

$$\frac{\partial \theta c_3}{\partial t} = \frac{\partial}{\partial z} \left(\theta D_3 \frac{\partial c_3}{\partial z} \right) - \frac{\partial q c_3}{\partial z} + k_{\text{nit}} \theta c_2 + k_{\text{nit}} \rho s_2 - k_{\text{den}} \theta c_3 \quad (7)$$

c_1 , c_2 , c_3 : concentrations of soluble Org-N, soluble $\text{NH}_4\text{-N}$, and soluble $\text{NO}_3\text{-N}$ (mgN cm^{-3}),
 s_1 , s_2 : concentrations of adsorbed Org-N and adsorbed $\text{NH}_4\text{-N}$ (mgN g^{-1}), ρ : bulk density (1.55g cm^{-3}), D_1 , D_2 , D_3 : dispersion coefficients of soluble Org-N, soluble $\text{NH}_4\text{-N}$, soluble $\text{NO}_3\text{-N}$ ($\text{cm}^2 \text{h}^{-1}$), k_{\min} , k_{nit} , k_{den} : the first-order reaction coefficients of mineralization,

nitrification, and denitrification (h^{-1})

The dispersion coefficient is formulated by

$$\theta D = D_L |q| + \theta D_w \tau \quad (8)$$

D_L : longitudinal dispersivity (cm), $|q|$: the absolute value of Darcy flux (cm h^{-1}), D_w : molecular diffusion coefficient in free water (cm^2h^{-1}), τ : tortuosity (-)

D_w of Org-N, $\text{NH}_4\text{-N}$, and $\text{NO}_3\text{-N}$ are set up to be zero because the nitrogen fluxes by diffusion are negligible compared to the dispersive transport. D_L is assumed to be 20cm.

For lack of better data, we assume that the first-order coefficients of soluble nitrogen and adsorbed nitrogen are identical. The isotherms of Org-N and $\text{NH}_4\text{-N}$ are formulated by a linear adsorption isotherm:

$$s_1 = k_{\text{org-N}} c_1 \quad s_2 = k_{\text{NH}_4\text{-N}} c_2 \quad (9)$$

Mineralization, nitrification, and denitrification are all linked to microbial activity, which is strongly temperature dependent. Lower soil temperature during the winter months significantly reduce microbial activity and, hence, transformation rates. We assumed that the soil surface temperature is 0°C during the periods from November 1st to March 31st every year and 20°C during the rest periods and calculated the heat transport equation,

$$C(\theta) \frac{\partial T}{\partial t} = \frac{\partial}{\partial z} \left(\lambda(\theta) \frac{\partial T}{\partial z} \right) - C_w q \frac{\partial T}{\partial z} \quad (10)$$

C : volumetric heat capacity of soil ($\text{J K}^{-1}\text{cm}^{-3}$), T : soil temperature, λ : thermal conductivity of soil ($\text{W cm}^{-1}\text{K}^{-1}$), C_w : volumetric heat capacity of water ($\text{J K}^{-1}\text{cm}^{-3}$) where λ is set to be a relatively high value so that soil temperature profile changes rapidly at the two switching dates with an accompanying rapid switch in reaction coefficients. Transformation rates are maximal during the summer and set to be negligible during the winter (when the model assumes soil temperatures of 0°C). These conditions only approximate actual conditions, but are considered appropriate in light of the lack of data on actual transformation rates during the winter. Not enough data existed to calibrate summer and winter time transformation rates separately.

H.2.3 Root Uptake of Water and Soluble NH_4-N and NO_3-N

Feddes' model is used to simulate root water uptake (Simunek et al., 1998). The model parameters are provided in the database in HYDRUS-1D for the vegetative period of corn and small grains. Furthermore, we consider the root growth modeled by Verhulst-Pearl's logistic growth function:

$$L_R(t) = f_r(t)L_m \quad (11)$$

L_R : root length, L_m : the maximum length of root, f_r : the root growth coefficient

L_m are set to be 91.44cm for corn and 60.96cm winter crop.

$$f_r(t) = \frac{L_0}{L_0 + (L_m - L_0)e^{-rt}} \quad (12)$$

L_0 : the initial root depth (set to be 0.01cm for both crops), r : the growth rate (h^{-1})

The growth rate r is calculated based on the assumption that 50% of the rooting depth will be reached after 50% of the growing season has elapsed.

The root distribution function is expressed by an exponential function with a maximum at the soil surface.

$$b(z) = ae^{-a(L-z)} \quad (13)$$

a : empirical constant (cm^{-1}) (set to be 0.105 for both crops), L : the z -coordinate of the soil surface (cm)

We assume that both crops only take up NH_4-N and NO_3-N (no organic N), that uptake is passive and that it is not limited by high concentrations (The maximally allowable concentrations are arbitrarily set to be 100,000 mg N cm^{-3} for NH_4-N and NO_3-N).

H.2.4 The partition of the calculated period initial conditions

Though we should consider two crops (corn and winter crop), HYDRUS-1D can deal with only one crop per simulation. Hence, we divided the total simulation period into seven individual simulation periods that were run sequentially:

Corn 98 :	1998/4/28 – 9/14	(140days)
Winter 99 :	1998/9/15 – 1999/4/26	(224days)

Corn 99 :	1999/4/27 – 10/11	(168days)
Winter 00 :	1999/10/12 – 2000/5/11	(213days)
Corn 00 :	2000/5/12 – 9/30	(142days)
Winter 01 :	2000/10/1 – 2001/5/18	(230days)
Corn 01 :	2001/5/19 – 9/21	(126days)

The initial conditions for Winter 99 to Corn 01 simulations are set to the terminal (final) conditions from the previous simulation period. The initial conditions for the first simulation (Corn 98) are the linearly interpolated values of measured data (volumetric water content, NH₄-N concentration, and NO₃-N concentration) on April 27th, 1998. The initial org-N concentration is set to be zero (Corn 98).

The simulated unsaturated domain includes the soil profile (root zon and the deeper vadose zone and stretches from the soil surface to the water table at 3m depth. The entire profile is assumed to be hydrologically homogeneous. The bottom boundary (3m depth) is defined as a water table boundary condition ($h=0$).

H.2.5 Boundary conditions at the soil surface

(1) Rainfall, Evaporation, and Transpiration

We use CIMIS (California Irrigation Management Information System) data at Modesto (Station #71) for rainfall (Figure 1) and potential evapotranspiration (Figure 3). The product of potential evapotranspiration and crop coefficient (UCCE, 1987) calculates the crop evapotranspiration (Figure 3).

The crop evapotranspiration (ET) is separated into evaporation (E) and transpiration (T) by the following equation (Campbell, 1985),

$$T/ET = 1 - \exp(-0.86LAI) \quad (14)$$

LAI : leaf area index (set to be 2.0, E / T = 0.25)

The crop coefficient is characterized by five time points (A, B, C, D, and E) that are indicated in UCCE (1987) (Table 1):

- A : planting
- B : 10 percent ground shading
- C : 75 percent or peak ground shading
- D : leaf aging effects on transpiration
- E : end of season

The actual planting and harvesting date are given for A and E, respectively. The values of B, C, and D are corrected by the proportional division (Figure 2).

(2) Irrigation and Fertilization Rates

The amounts of irrigation, Org-N and NH₄-N concentrations in manure, and chemical nitrogen were measured on most occasions (Figure 4). Missing data are replaced using the following procedure.

Preirrigation amounts on 1998/5/4 are assumed to be averages of preirrigation amounts for corn in 1999 and 2000 (1999/6/12 and 2000/5/6). NH₄-N and Org-N amounts on 1998/5/4 are assumed to be averages of concentrations of the first applied lagoon water in summer in 1999, 2000 and 2001 (1999/7/14, 2000/6/14 and 2001/7/3). Preirrigation for winter crop in 1998 was conducted until 1998/9/10 (personal communication, M. Campbell-Mathews). Planting date of winter crop in 1998 was 1998/9/22. So, this irrigation date is assumed to be 1998/9/10. Irrigation amounts on this date are assumed to be averages of the first irrigation for the winter crop in 1999, 2000 and 2001 (1999/11/1, 2000/10/23 and 2001/10/17). Irrigation amounts on 1998/10/22, 1999/3/1, 2000/3/1 and 2001/4/17 are assumed to be the average of measured irrigation amounts for winter crop in 1999, 2000 and 2001.

The source of applied NH₄-N for corn in 1998 (1998/6/20-1998/8/6), on 1999/3/1 and 2000/3/1 was commercial fertilizer (anhydrous ammonia). The source of the nitrate was the fresh irrigation water in the canal. The fresh water source is from mountain snowmelt stored in reservoirs then supplemented by varying amounts of pumped shallow groundwater. Nitrate is mostly from groundwater, hence nitrate concentrations vary from irrigation to irrigation. Missing values (red in Table 2) are due to missing samples of the canal water or missing measurements, or both. Missing values are assumed to be the average of measured nitrate concentrations before and after their respective date (see Figure 5).

H.3 Calibration and Results

H.3.1 Calibration of Hydraulic Properties of Soil

The parameters of van Genuchten's equations (α , n , θ_r , θ_s , K_s) are unknown. We assumed that θ_s is equal to porosity set to be 0.404 because the measured bulk density is 1.55g cm⁻³ and the

soil particle density is assumed to be 2.6. The rest of the parameters are determined by inverse solution (calibration). Calibration of these hydraulic parameters was done separately for three different calibration periods:

Case 1: Corn 98 (1998/4/28 – 1998/9/14, Target: 40 soil water content data)

Case 2: Corn 99 (1999/8/9 – 1999/9/12, Target: 72 groundwater level data at a well near the field)

Case 3: Corn 01 (2001/8/2 – 2001/9/21, Target: 15 soil water content data including measurements before and after irrigation)

The results of the inverse solution are shown in Table 3. The results of the inverse solution are summarized as follows:

Case 1: Figure 7 indicates that the simulation of soil water content is very good for the calibration period (1998), but also for the validation years 1999 and 2000. Only in 2001, the soil water content predicted from this calibration was not adequate.

Case 2: This calibration, which did not use soil water content but groundwater level data for 1999 yielded parameters and validation results similar to Case 1 (Figure 8). However, calibration against groundwater levels was not satisfactory (Fig.10). The calibrated saturated hydraulic conductivity is 100cm d^{-1} which is the same as the minimum value specified for the calibration indicating convergence problems during the calibration. The laboratory measured value for the saturated hydraulic conductivity is 1358cm d^{-1} using the falling head method. This calibration should be rerun in future work by combining the unsaturated model with a groundwater model that can also account for the lateral flow of water at and below the water table. Lateral groundwater flow was not considered in this calibration.

Case 3: Figure 9 shows that this calibration provides a good fit to the measured 2001 data used for the calibration, but also for the (predicted) validation periods 1998, 1999, and 2000. The parameter set obtained from the calibration is consistent with the measured soil texture, classified as loamy sand according to grain size analysis (sand : 81%, silt : 15%, clay : 4%). Calibrated soil water retention curves from the Case 3 calibration match best with typical retention curves for loamy sand (Figure 6). Thus, we use the calibrated parameters obtained from the Case 3 calibration for the transport calibration and predictive work in the following sections.

H.3.2 Isotherm Coefficient of $\text{NH}_4\text{-N}$

The isotherm coefficient of $\text{NH}_4\text{-N}$ is estimated by calibrating to plant N uptake data and is estimated to be 0.52 (Figure 11).

H.3.3 Sensitivity Analysis

The isotherm coefficient of Org-N ($k_{\text{Org-N}}$) and the first-order reaction coefficients of mineralization (k_{min}), nitrification (k_{nit}), and denitrification (k_{den}) are unknown. These parameters are determined by calibration via an extensive sensitivity analysis. For the sensitivity analysis, sequential simulations of the seven crop periods were performed for a total of 1470 parameter combinations: 7 different $k_{\text{Org-N}}$ levels, 5 different k_{min} levels, 6 different k_{nit} levels, and 7 different k_{den} levels. All possible combinations were simulated and root mean square errors (RMSE) mapped across the mineralization-nitrification parameter space (Figure 12a-g). A set of six maps were generated for each of the six denitrification rates. Seven set (pages) of maps were generated, one for each of the seven sorption coefficients for organic nitrogen. Transformation rates are reported in units of $[\text{days}^{-1}]$.

On all maps, the least RMSE (best simulations) is observed in the upper left map, where the denitrification rate is zero (no denitrification). With non-zero denitrification rates, the predicted results deviated from measured data. Errors (RMSE) become larger with larger denitrification rates. It is therefore likely that denitrification plays a negligible role.

Within each of the maps, including the upper left maps, the least RMSE is observed around $\log(k_{\text{nit}}) \sim -1$ and is relatively insensitive to $\log(k_{\text{min}})$ values above -2. Very similar patterns are observed across all sorption rates for organic N, k_{rgN} . However, the minimum RMSE observed within individual maps for $k_{\text{den}} = 0$ is lower as the sorption rates increases. Hence, the overall optimal result (best case) are obtained for large organic N sorption $k_{\text{Org-N}}=1000$, relatively fast mineralization rates $k_{\text{min}}\sim 1.6$, rapid nitrification, $k_{\text{nit}}=0.1$, and no denitrification $k_{\text{den}}=0$. The largest RMSE (worst parameter set) at high organic N sorption is obtained for $k_{\text{Org-N}}=10000$, very fast mineralization and nitrification rates $k_{\text{min}}=100$, $k_{\text{nit}}=0.0001$, and high denitrification rates $k_{\text{den}}=1$.

To illustrate the sensitivity of the predictions to this wide parameter range, ammonium and nitrate profiles and their transient behavior during 1998-2001 are shown for both the best case (Figure 13-14) and worst case (Figure 15-16) scenario. Clearly, the worst case parameter set provides unacceptable results. Hence, relatively high significance can be given to the best case parameter set, at least to the order of magnitude of the individual parameters. Modeling results are more sensitive to nitrification rates (approximately 0.1 d^{-1}), but much less sensitive to mineralization (anywhere in the range of $0.1 \text{ d}^{-1} - 100 \text{ d}^{-1}$). This would support the

conclusion that the vast portion of the organic nitrogen applied mineralizes very quickly – with half-lives of a few tens of days at the most, perhaps even shorter.

With the optimized parameters, we simulated the nitrogen fluxes across the water table. All of the nitrogen at the water table is in form of nitrate. The total groundwater loading over the seven cropping seasons is nearly 15 mg N cm^{-2} (1500 kg N / ha) (Figure 17), approximately the same as the cumulative crop nitrogen uptake (Figure 18). Nitrogen leaching decreased after 1998 due to improved nutrient management practices (Figure 18, Figure 20), while plant nitrogen uptake continued at the same rate throughout 1998-2001. Cumulative mineralization and nitrification are shown in Figure 19, which reflects the model assumption of zero transformation during the winter months. It is conceivable that allowing for some transformation during the winter months (e.g., at 25% of the summer rate) would impact the calibration results towards slower overall mineralization rates. This issue should be pursued in future research. Clearly, though, mineralization occurs rapidly during the summer months.

Figure 21 illustrates the simulated nitrogen concentration time-line of the recharge water at the water table (bottom of the simulated domain). Recharge concentrations initially are 50 mg N/l , then decrease to approximately 25 mg/l . This agrees well with results from groundwater monitoring after accounting for the travel time between the field and monitoring wells. Higher concentrations in the winter 2000 are due to large amounts of mineralized (and nitrified) nitrogen leaching to the water table after the main growing season in 1999. Concentration is not a measure of nitrogen fluxes, however. It merely indicates the nitrate concentration at the bottom of the unsaturated zone profile. Actual daily loading rates to groundwater (in units of $\text{kg N ha}^{-1} \text{ yr}^{-1}$) are shown in Figure 22. The results indicate that groundwater loading is a very transient event with high spikes associated with high irrigation amounts or significant precipitation events.

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Table 1: Parameters of reference crop coefficients of corn and small grains (winter crop) in the San Joaquin Valley (UCCE, 1987).

	Corn	Small grains
A	5/16	11/1
B	6/7	12/14
C	7/16	1/25
D	8/29	3/25
E	9/30	5/15
Kc1	0.19	0.25
Kc2	1.06	1.20
Kc3	0.55	0.40

Table 2: Measured irrigation volume and measured NH₄-N, Org-N and NO₃-N concentrations at the field. Shaded values or dates are estimated.

		Bun avg	Bun1west	Bun2cente	Bun3east	Bun avg	Bun1west	Bun2cente	Bun3east	Bun avg	Bun1west	Bun2cente	Bun3east	Bun avg
	Irrigation					NH4-N				Org-N				NO3-N
	ac-in/A					Lb/A				Lb/A				Lb/A
1998/5/4	pre	6.4	6.3	6.5	?	33	29	38	31	42	42	56	27	4.5
1998/6/20	1	3.1	3.4	2.9	2.9	33	48	27	18	0	0	0	0	1.3
1998/7/1	2	3.4	3.6	3.3	3.3	31	49	27	15	0	0	0	0	4.5
1998/7/11	3	3.7	4.0	3.5	3.5	62	66	68	52	0	0	0	0	4.5
1998/7/19	4	4.1	3.9	4.2	4.2	53	60	60	22	0	0	0	0	4.5
1998/7/28	5	4.0	3.1	4.5	4.5	0	0	0	0	0	0	0	0	4.5
1998/8/6	6	3.5	3.1	3.7	3.7	45	46	58	30	0	0	0	0	4.5
1998/8/15	7	4.5	4.6	4.3	?	0	0	0	0	0	0	0	0	4.5
1998/8/24	8	4.5	4.4	4.6	?	0	0	0	0	0	0	0	0	4.5
1998/9/10	pre	6.8	7.6	6.5	6.3	0	0	0	0	0	0	0	0	4.5
1998/10/22		7.3	7.5	7.3	7.0	0	0	0	0	0	0	0	0	4.5
1999/3/1		7.3	7.5	7.3	7.0	80	80	80	80	0	0	0	0	4.5
1999/6/12	pre	8.3	7.8	8.8	8.3	0	0	0	0	0	0	0	0	4.5
1999/7/14	1	3.7	3.9	3.9	3.3	33	27	48	24	95	92	134	59	2.1
1999/7/24	2	5.5	4.2	7.2	5.1	44	67	16	50	54	71	36	55	4.7
1999/8/2	3	5.3	4.3	5.9	5.8	52	45	56	57	52	41	38	77	2.2
1999/8/13	4	3.8	3.8	3.6	4.1	53	55	38	66	25	24	17	35	4.5
1999/8/24	5	6.2	5.3	6.7	6.6	52	50	53	51	31	29	33	32	4.5
1999/9/1	6	5.6	4.5	6.5	5.7	0	0	0	0	0	0	0	0	4.5
1999/9/11	7	5.4	5.1	5.7	5.3	0	0	0	0	0	0	0	0	4.5
1999/11/1		7.4	8.7	7.5	6.0	0	0	0	0	0	0	0	0	4.5
2000/3/1		7.3	7.5	7.3	7.0	80	80	80	80	0	0	0	0	4.5
2000/4/1		7.2	4.1	10.3	7.3	0	0	0	0	0	0	0	0	4.5
2000/4/27		7.0	5.9	6.3	8.9	0	0	0	0	0	0	0	0	4.5
2000/5/6	pre	4.5	4.7	4.3	?	0	0	0	0	0	0	0	0	4.8
2000/6/14	1	6.8	5.1	6.7	8.7	41	37	43	41	16	18	17	13	8.3
2000/6/22	2	5.3	5.0	5.7	5.3	43	41	34	55	15	18	10	16	2.4
2000/7/1	3	5.4	5.0	5.8	5.3	24	25	22	25	9	13	7	8	4.6
2000/7/11	4	4.3	4.4	4.3	4.4	41	19	43	61	16	5	22	20	4.9
2000/7/22	5	4.7	5.5	4.1	4.4	0	0	0	0	0	0	0	0	8.8
2000/7/31	6	4.2	4.2	4.6	3.9	17	16	17	16	16	12	12	23	4.6
2000/8/9	7	6.0	5.8	6.5	5.7	16	22	13	14	17	13	15	23	5.1
2000/8/19	8	5.9	5.3	6.3	6.3	0	0	0	0	0	0	0	0	3.1
2000/10/23		6.8	6.5	6.9	6.9	0	0	0	0	0	0	0	0	4.5
2001/2/26		6.1	6.2	6.1	6.1	182	187	180	180	71	72	70	70	8.1
2001/3/28		10.2	13.6	8.7	8.1	0	0	0	0	0	0	0	0	4.5
2001/4/17		7.3	7.5	7.3	7.0	0	0	0	0	0	0	0	0	4.5
2001/5/25	pre	11.9	14.5	14.5	6.8	0	0	0	0	0	0	0	0	4.4
2001/6/24	1	9.6	11.7	11.7	5.3	0	0	0	0	0	0	0	0	4.4
2001/7/3	2	6.2	6.8	6.8	4.9	25	23	23	28	14	17	17	8	4.3
2001/7/12	3	7.2	8.1	8.1	5.3	47	47	47	47	18	19	19	17	4.7
2001/7/22	4	7.1	7.6	7.6	6.0	39	38	38	40	17	15	15	21	3.4
2001/8/2	5	6.8	7.1	7.1	6.1	41	46	46	33	17	18	18	14	3.8
2001/8/11	6	5.5	5.8	5.8	4.9	0	0	0	0	0	0	0	0	4.0
2001/8/22	7	7.0	7.7	7.7	5.6	33	35	35	29	24	25	25	22	5.5
2001/9/2	8	6.7	6.8	6.8	6.4	0	0	0	0	0	0	0	0	4.5
2001/9/12	9	6.7	6.9	6.9	6.2	0	0	0	0	0	0	0	0	4.5
2001/10/17		6.2	7.5	5.2	5.9	0	0	0	0	0	0	0	0	4.5
Average		6.0	6.1	6.3	5.7	25.0	25.8	24.8	23.8	11.4	11.3	11.7	11.2	4.5
Average for winter crop		7.3	7.5	7.3	7.0									

Table 3: *Calibrated van Genuchten's parameters.*

	θ_r	θ_s	α	n	K_s (cm d ⁻¹)	RMSE ¹
Case 1 (Corn 98)	0.0033384	0.404	0.019215	1.326	603.13	0.053564
Case 2 (Corn 99)	0.030073	0.404	0.014028	1.4931	100.0	0.060972
Case 3 (Corn 01)	0.000921	0.404	0.029278	1.5026	922.2	0.069196

¹RMSE is root mean square error between the measured and the calculated volumetric water content during 1998 to 2001.

Table 4: *Nitrogen transformation parameters and the linear isotherm coefficient of Org-N for sensitivity analysis.*

k_{\min}	$t_{1/2\min}$	k_{nit}	$t_{1/2\text{nit}}$	k_{den}	$t_{1/2\text{den}}$	$k_{\text{Org-N}}$	$R_{\text{Org-N}}$
0.0001	6931.47	0.0001	6931.47	0	-	0.1	1.38
0.001	693.15	0.001	693.15	0.0001	6931.47	1	4.84
0.01	69.31	0.01	69.31	0.001	693.15	10	39.37
0.1	6.93	0.1	6.93	0.01	69.31	100	384.66
1	0.69	1	0.69	0.1	6.93	1000	3837.63
10	0.069			1	0.69	10000	38367.3
100	0.0069					100000	383664.3

Note on Table 4:

$$t_{1/2} = -\ln(0.5)/k_{\text{min, nit or den}} \quad R_{\text{Org-N}} = k_{\text{Org-N}} \rho/n + 1 \quad (15)$$

where ρ is bulk density, n is porosity, $t_{1/2}$ is the half life period, R is the retardation factor.

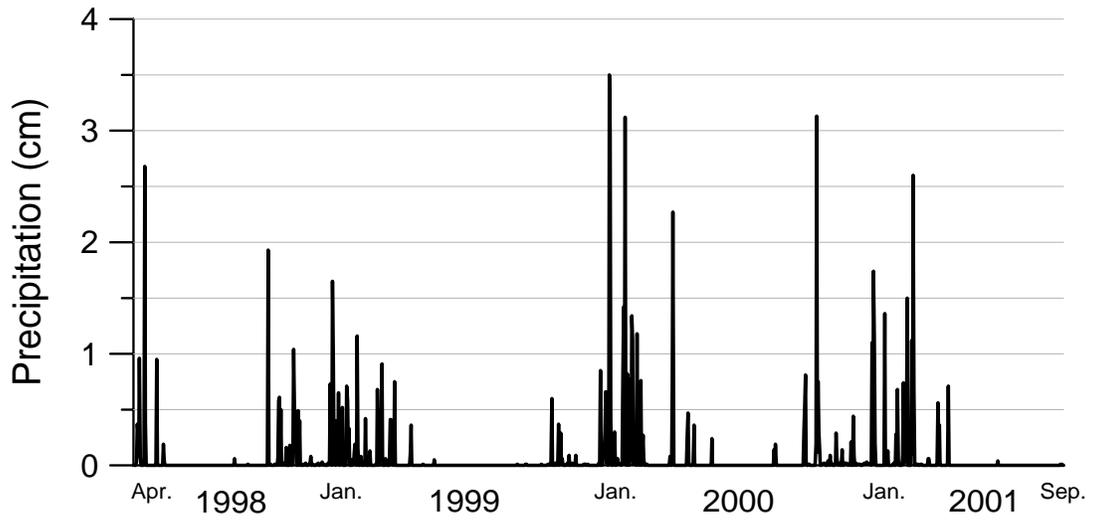


Figure 1: *Daily precipitation during the simulation period at Modesto CIMIS station.*

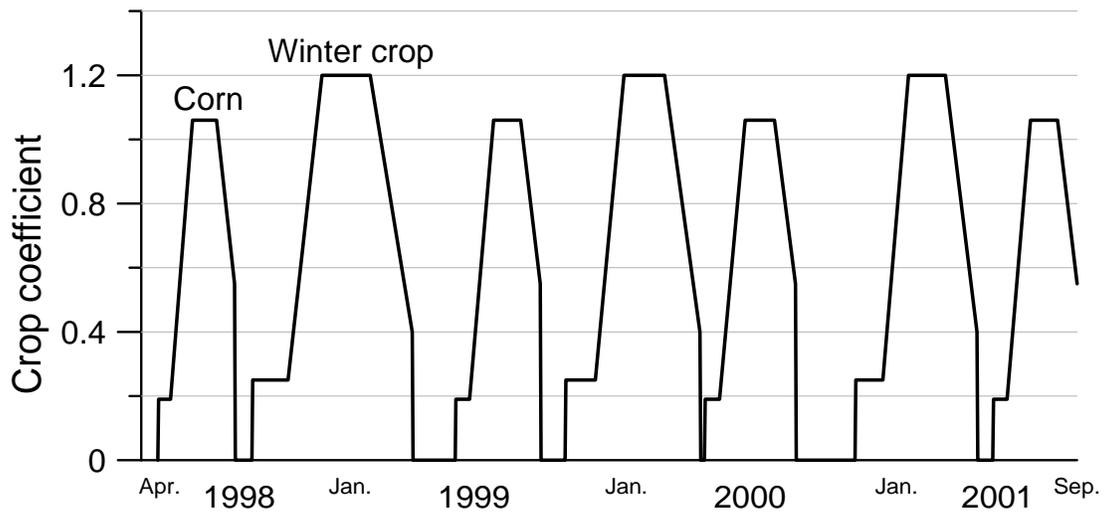


Figure 2: *Temporal change in crop coefficient.*

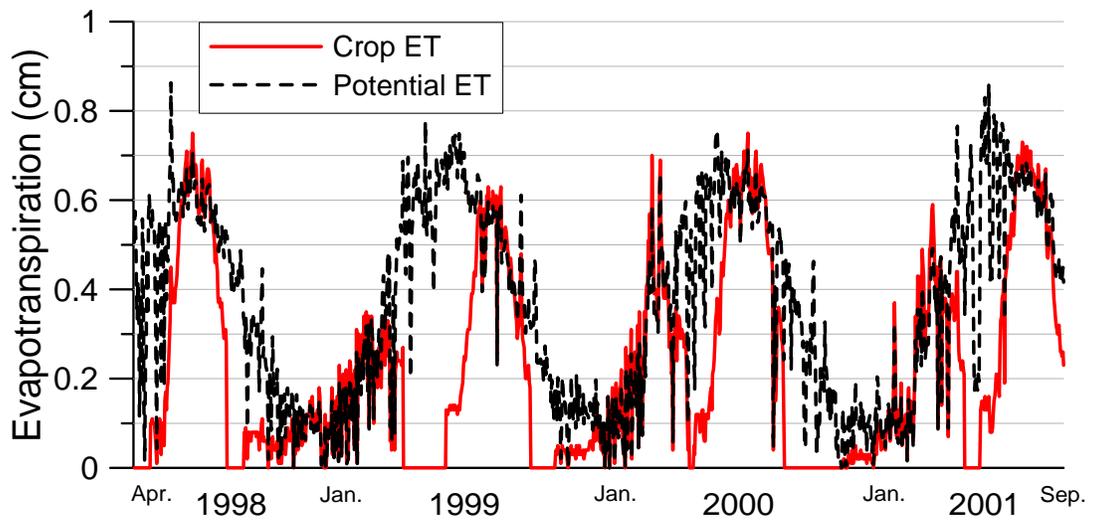


Figure 3: *Potential and crop evapotranspiration.*

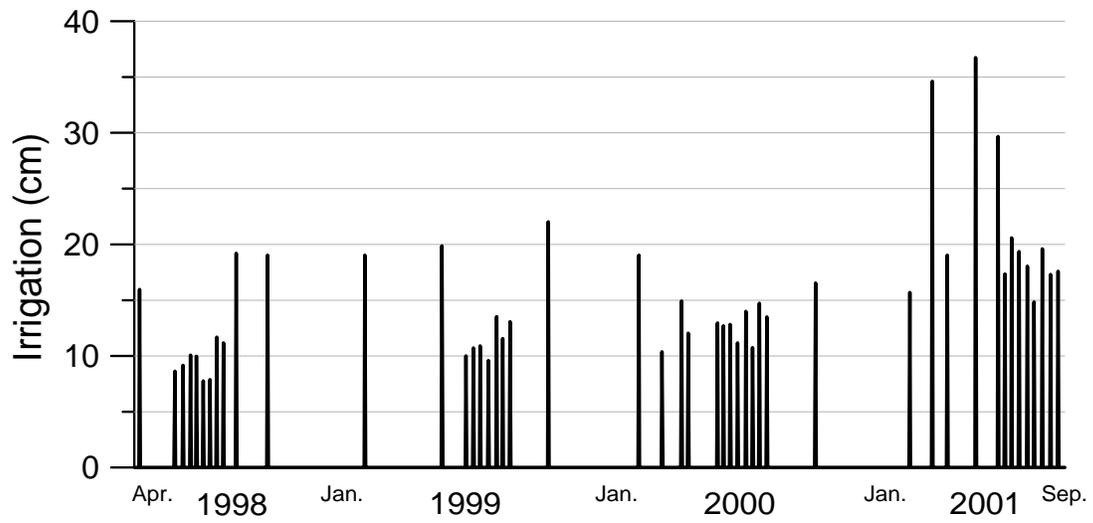


Figure 4: *Irrigation amount.*

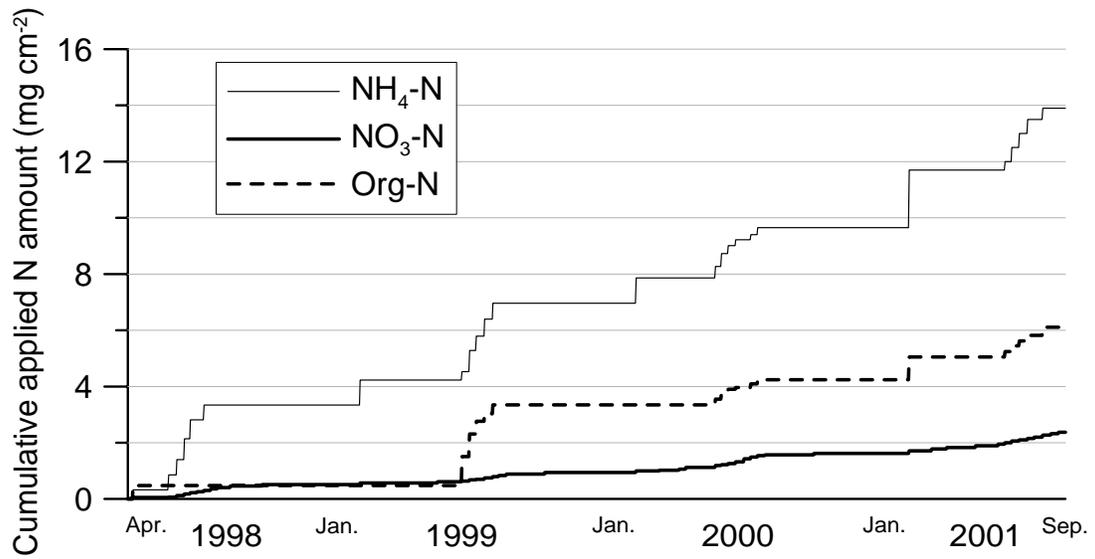


Figure 5: *Cumulative applied Org-N, NH₄-N and NO₃-N amounts at the field.*

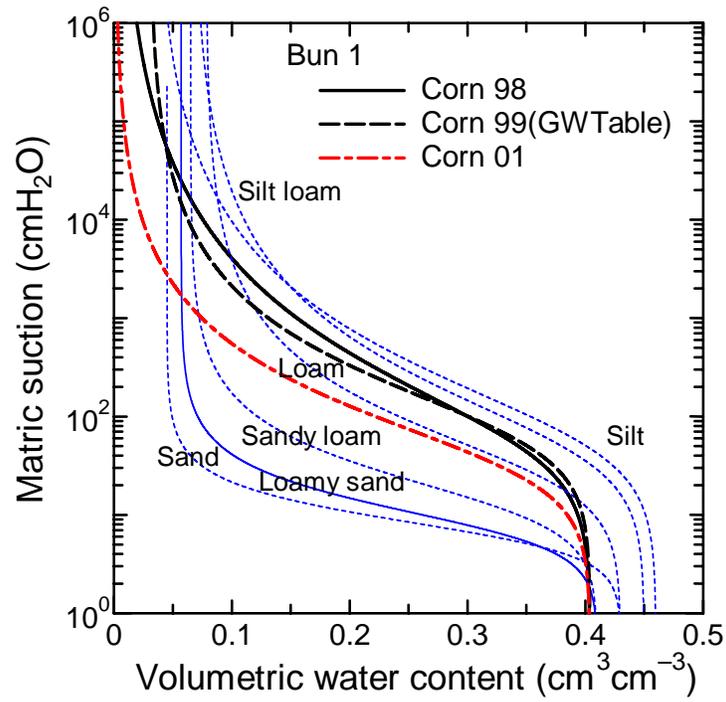


Figure 6: *Calibrated soil water retention curves (three heavy lines) and curves for sand, loamy sand, sandy loam, loam, silt, and silt loam based on HYDRUS-1D database.*

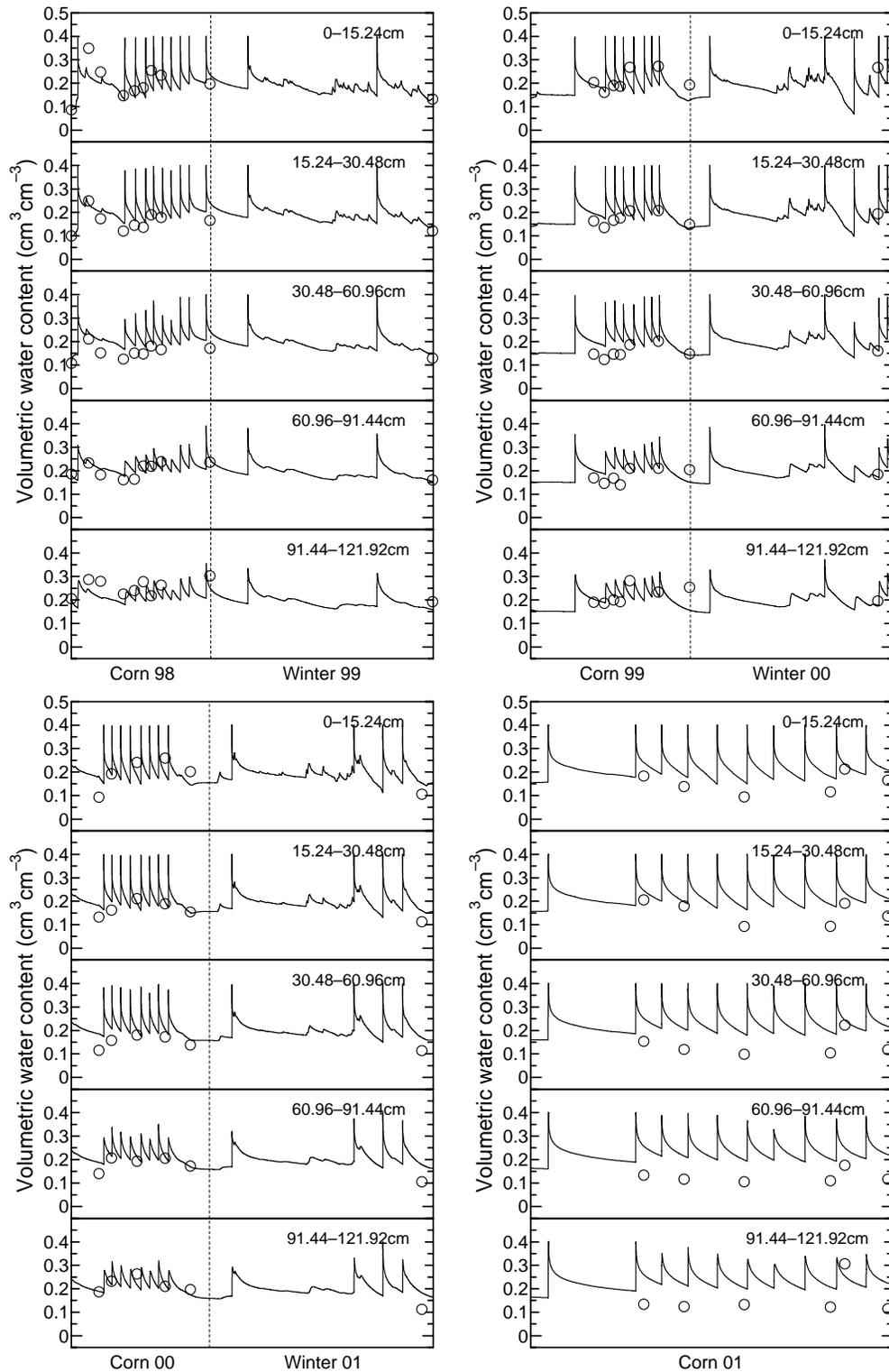


Figure 7: Measured and calculated temporal changes in volumetric water content in various soil depths. Symbols represent measured data and solid lines represent calculated changes based on the optimization of soil hydraulic properties for the Case 1.

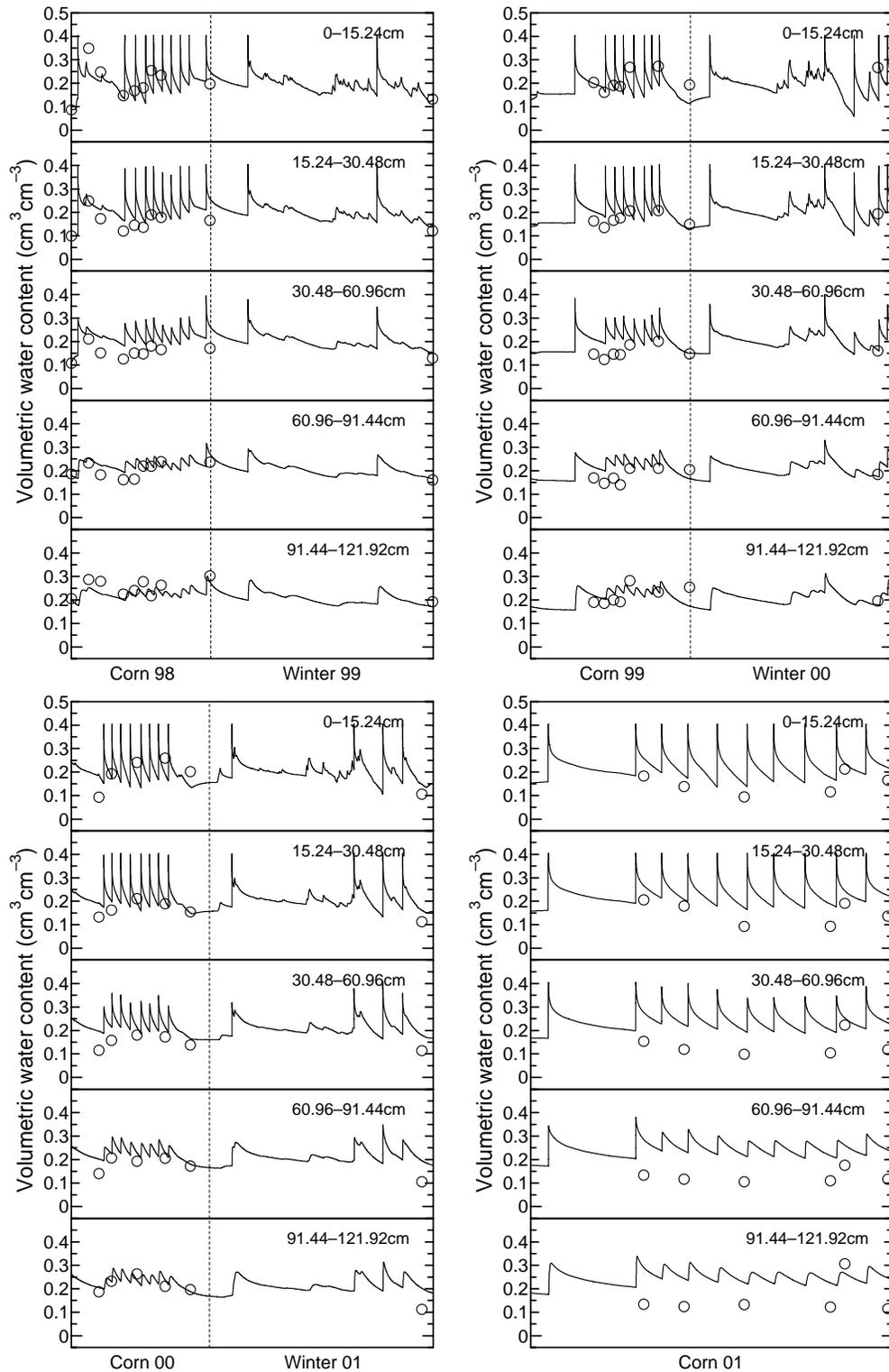


Figure 8: Measured and calculated temporal changes in volumetric water content in various soil depths. Symbols represent measured data and solid lines represent calculated changes based on the optimization of soil hydraulic properties for the Case 2.

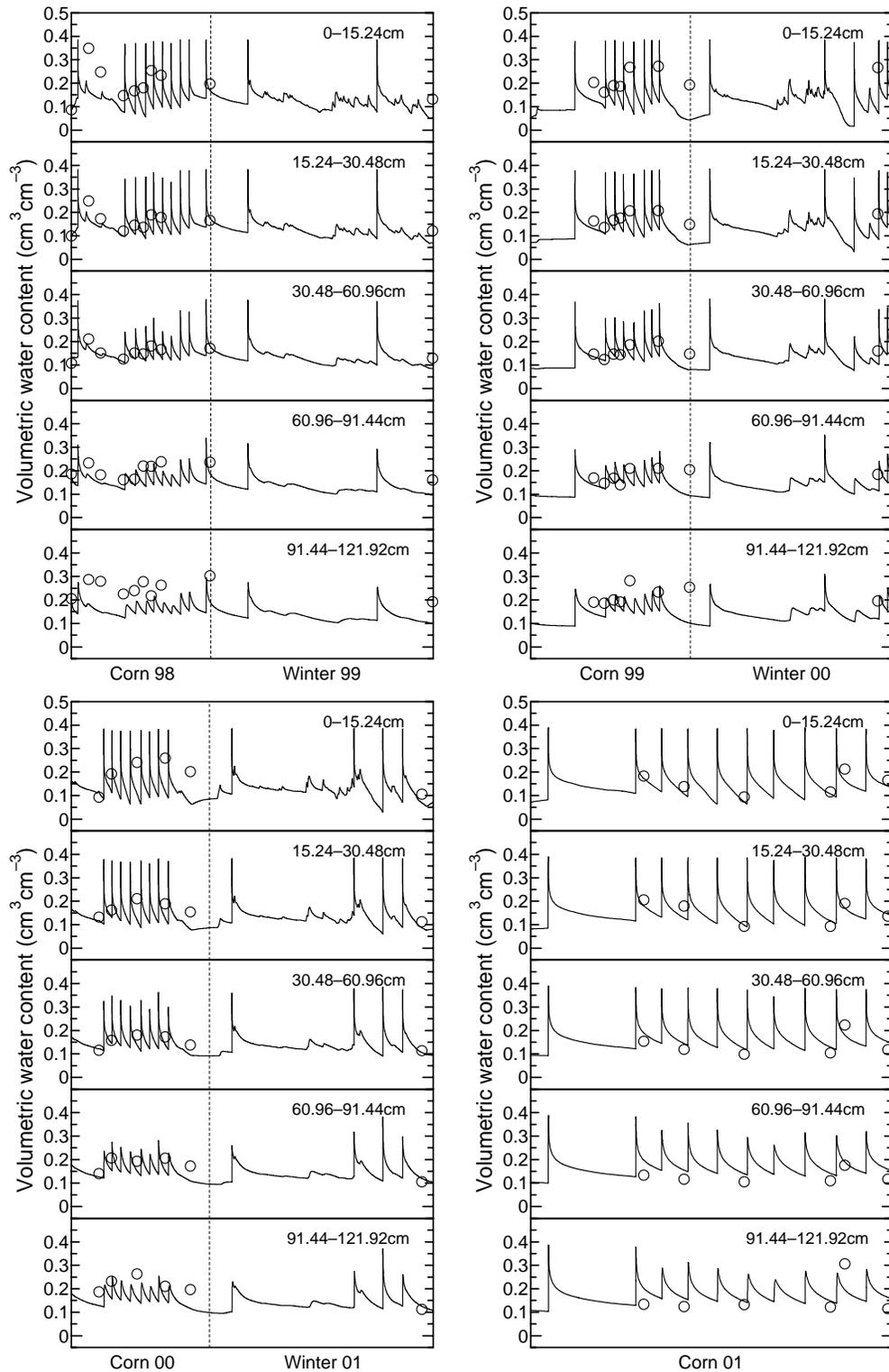


Figure 9: Measured and calculated temporal changes in volumetric water content in various soil depths. Symbols represent measured data and solid lines represent calculated changes based on the optimization of soil hydraulic properties for the Case 3.

Observation Nodes: Pressure Heads

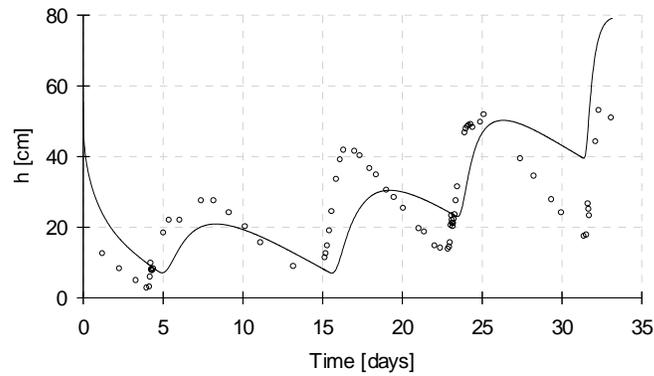


Figure 10: Measured and calculated targeted groundwater levels in the case of Case 2.

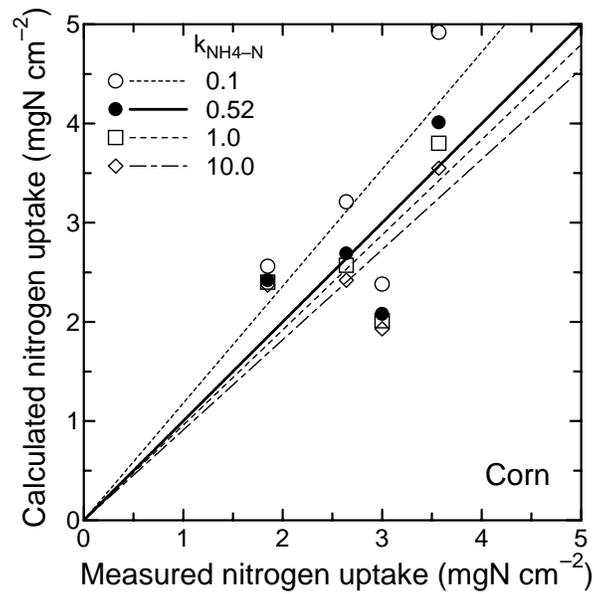


Figure 11: Measured and calculated N uptake amounts for various values of the $\text{NH}_4\text{-N}$ isotherm coefficient. N uptakes were calculated from soluble $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ in the root zone, which was calculated from the measured $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ and the assumed isotherm coefficient, and root water uptake fluxes estimated from water movement simulation.

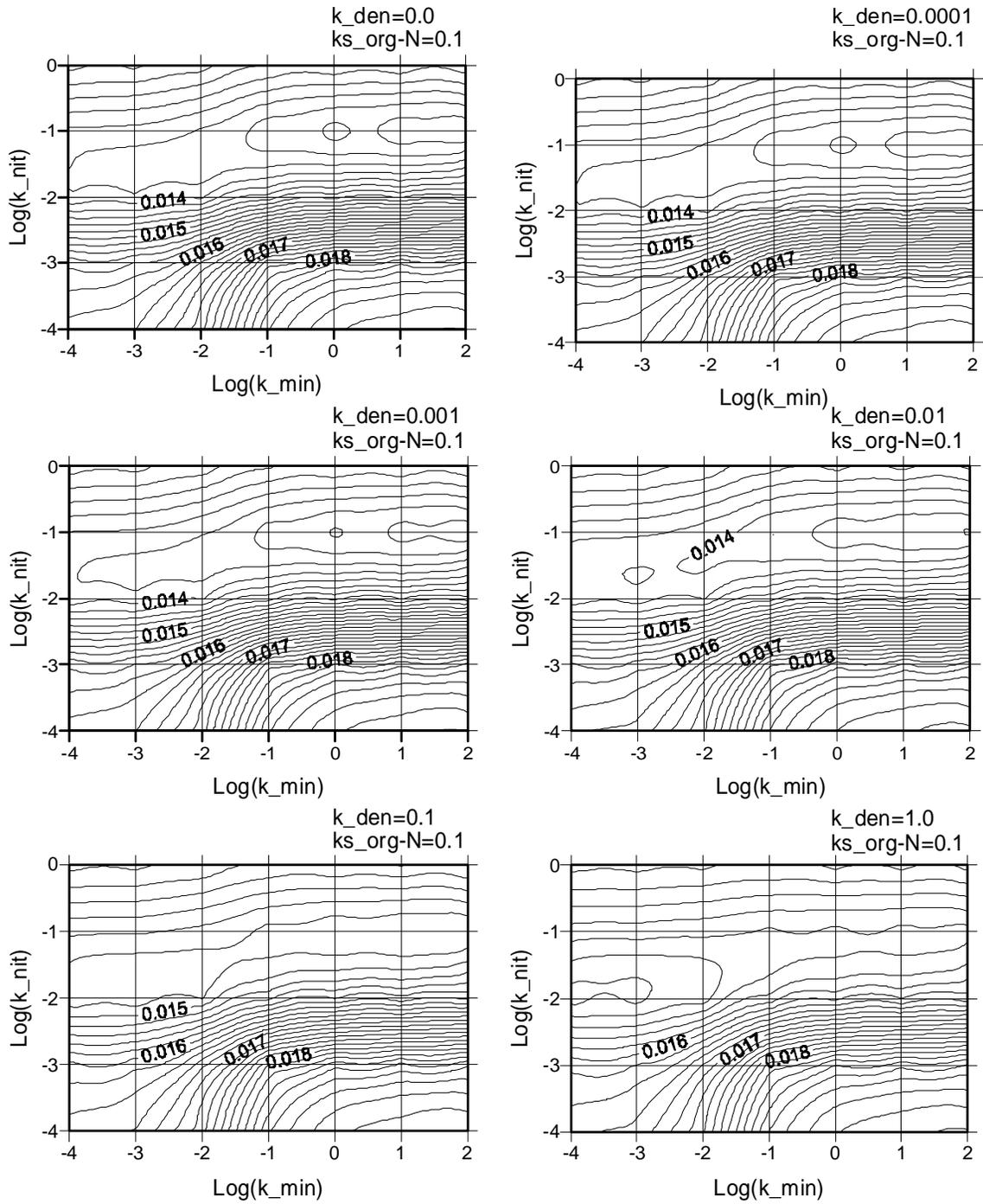


Figure 12(a): Contour maps of root mean square error between measured and calculated total $\text{NH}_4\text{-N}$ concentrations and $\text{NO}_3\text{-N}$ concentrations in soil ($k_{\text{Org-N}} = 0.1$).

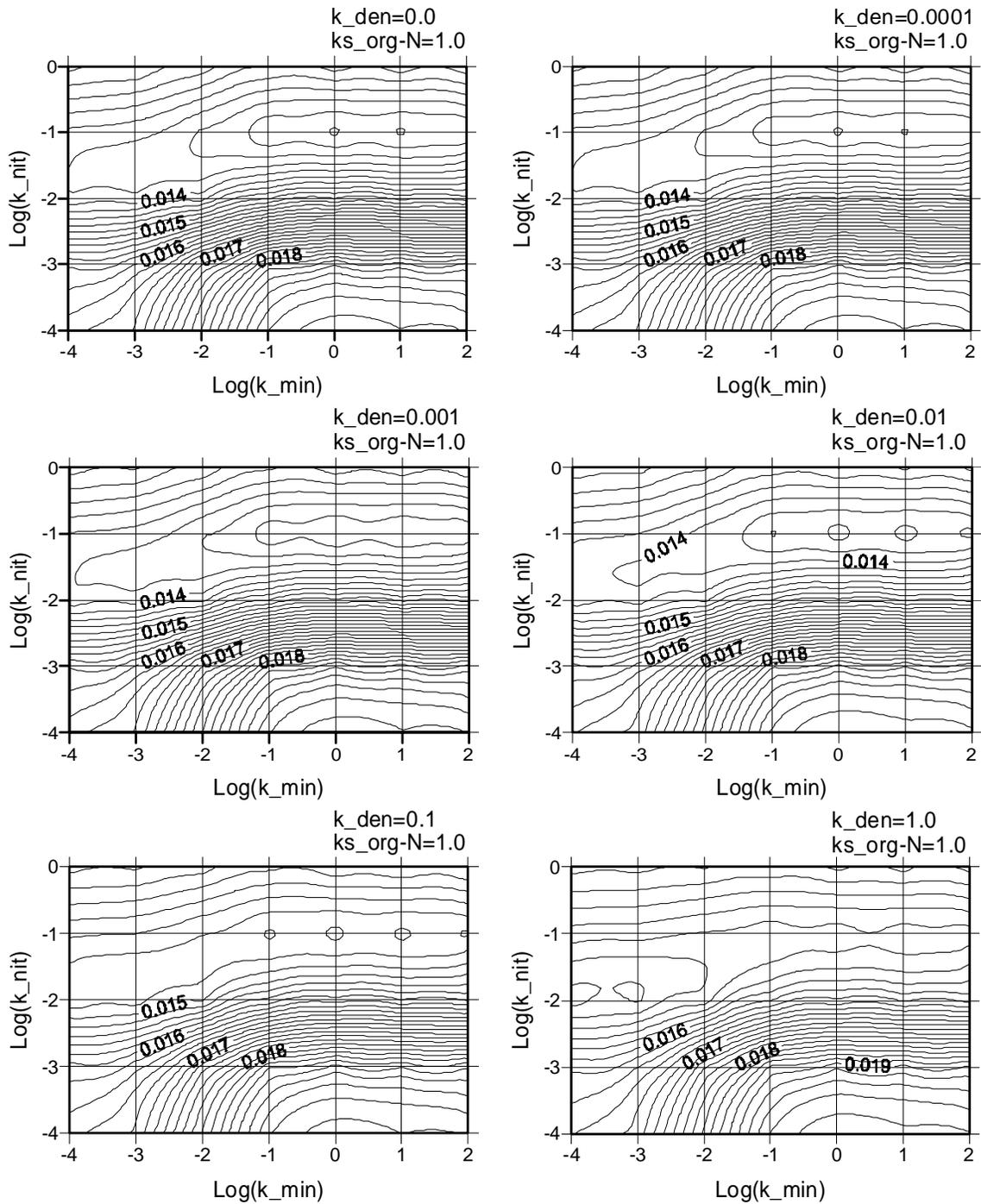


Fig.12(b): Contour maps of root mean square error between measured and calculated total $\text{NH}_4\text{-N}$ concentrations and $\text{NO}_3\text{-N}$ concentrations in soil ($k_{\text{Org-N}} = 1.0$).

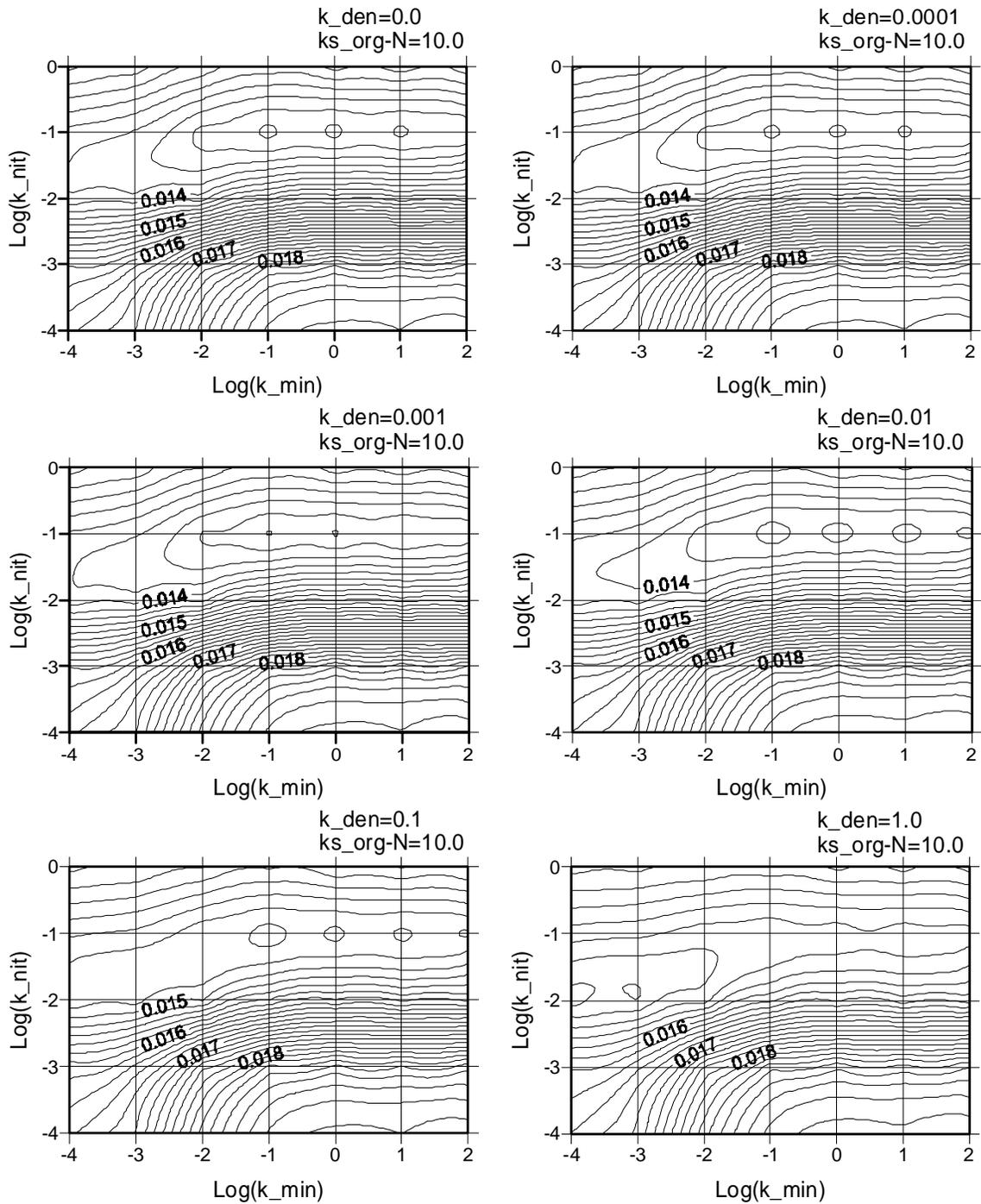


Fig.12(c): Contour maps of root mean square error between measured and calculated total NH_4-N concentrations and NO_3-N concentrations in soil ($k_{Org-N} = 10.0$).

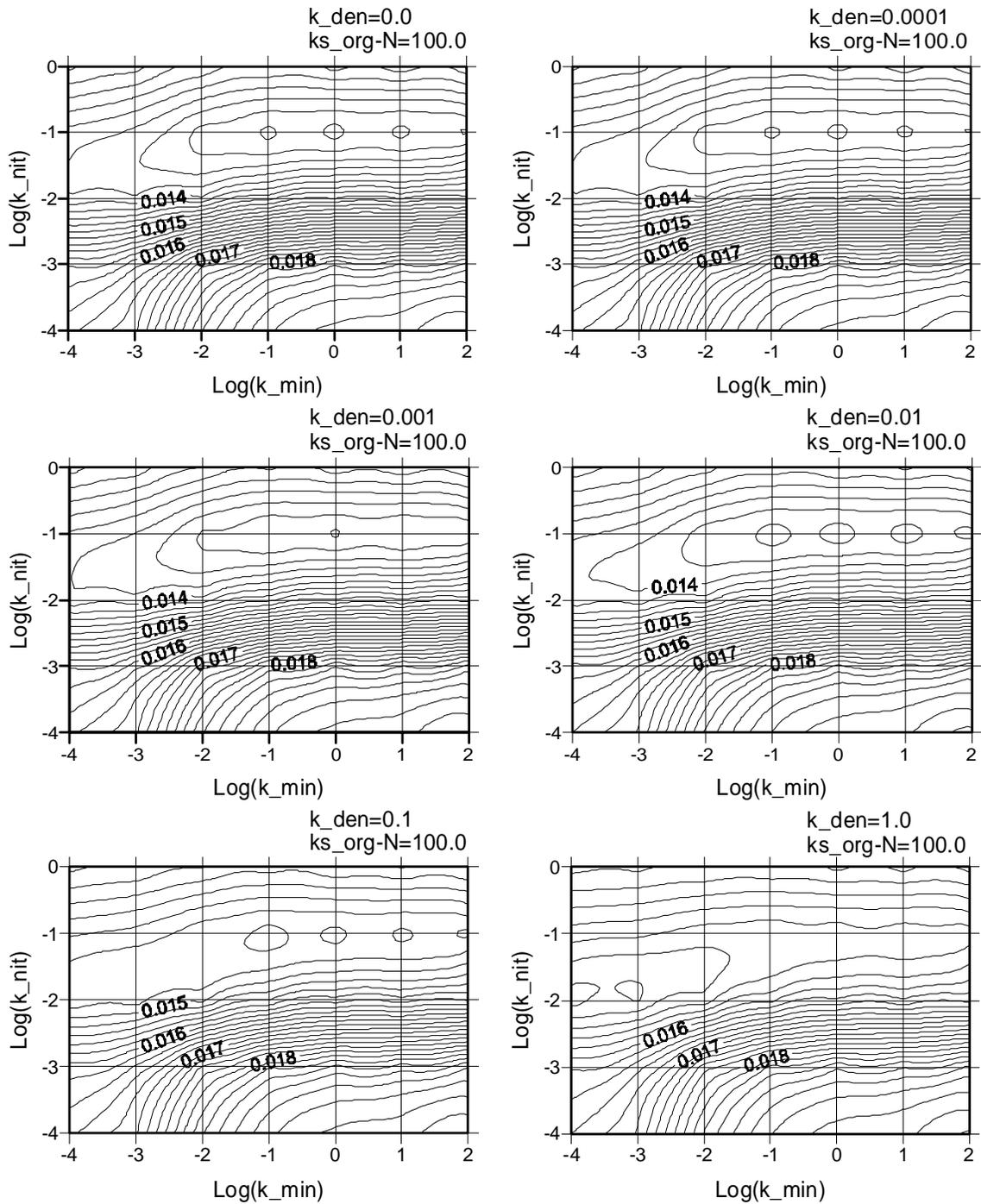


Figure 12(d): Contour maps of root mean square error between measured and calculated total $\text{NH}_4\text{-N}$ concentrations and $\text{NO}_3\text{-N}$ concentrations in soil ($k_{\text{Org-N}} = 100.0$).

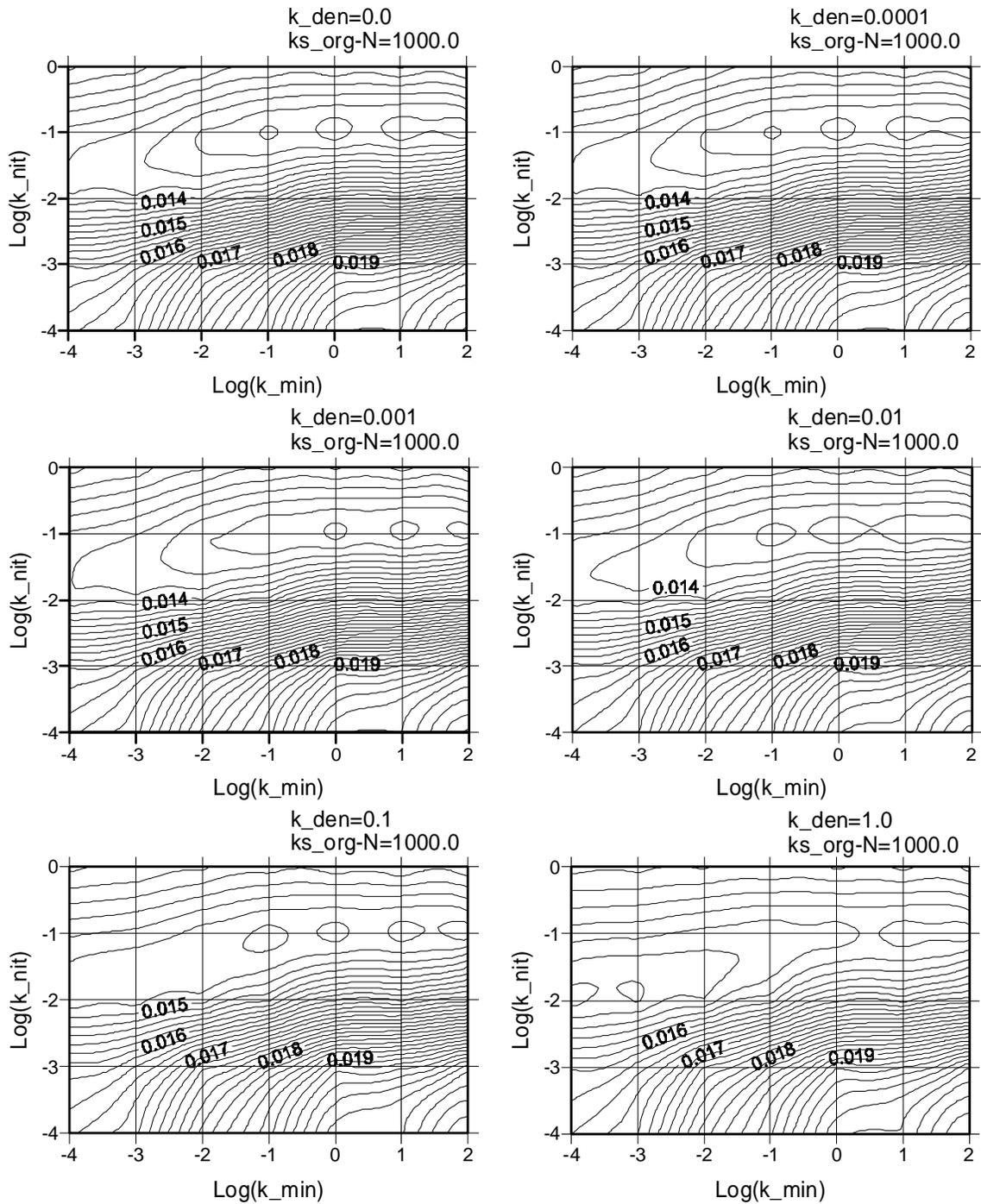


Figure 12(e): Contour maps of root mean square error between measured and calculated total $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ concentrations in soil ($k_{\text{org-N}} = 1000.0$).

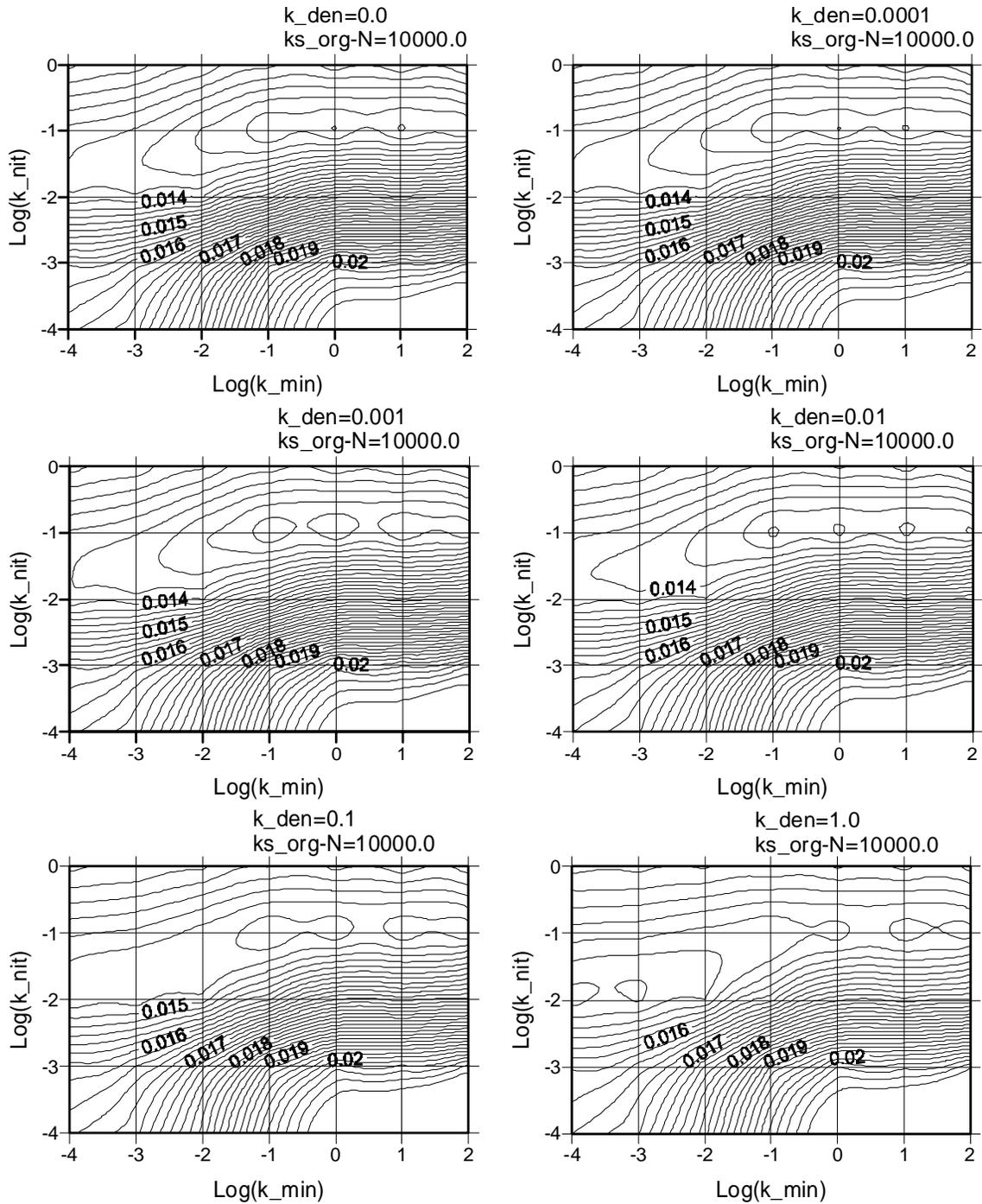


Figure 12(f): Contour maps of root mean square error between measured and calculated total $\text{NH}_4\text{-N}$ concentrations and $\text{NO}_3\text{-N}$ concentrations in soil ($k_{\text{Org-N}} = 10000.0$).

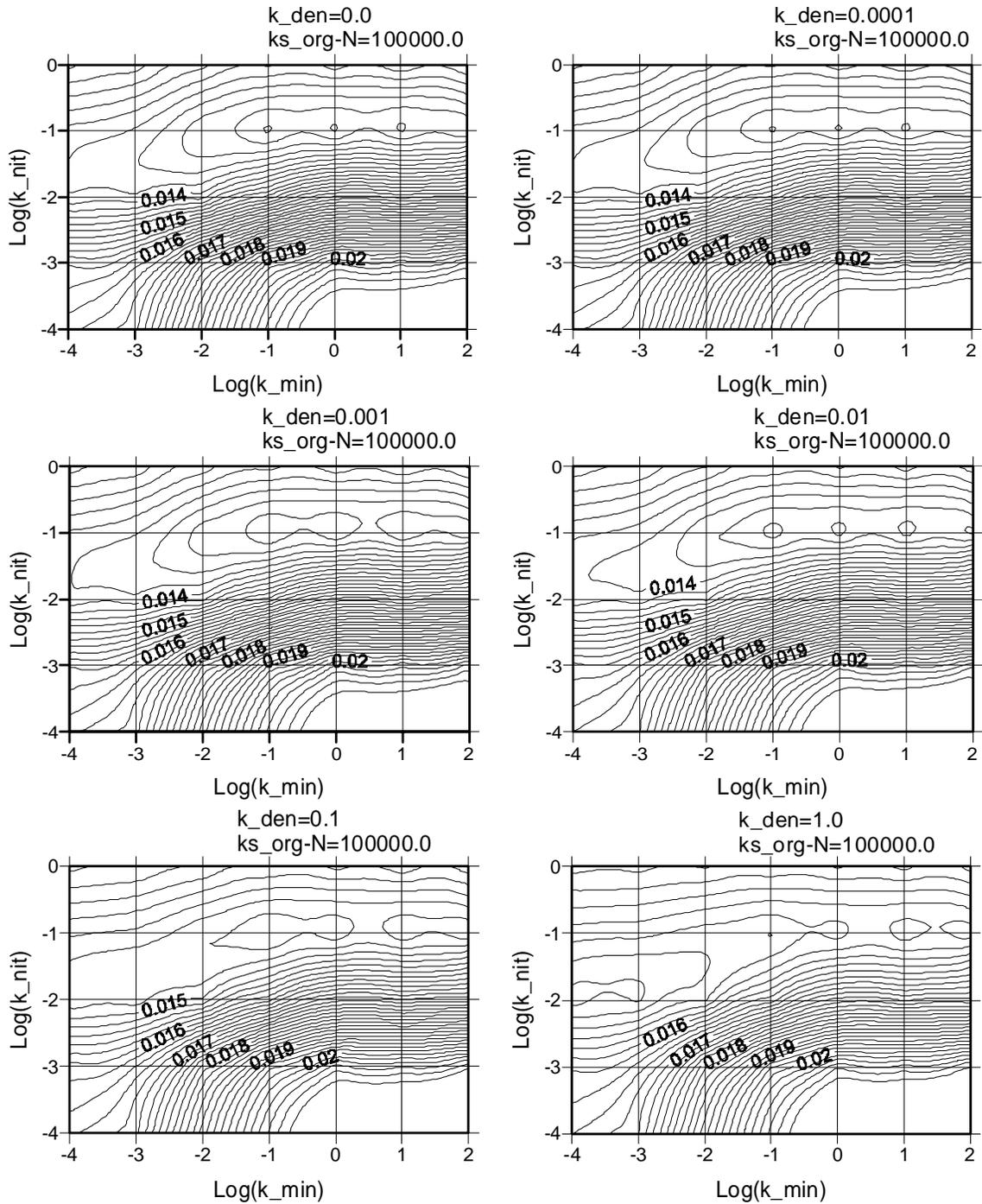


Figure 12(g): Contour maps of root mean square error between measured and calculated total $\text{NH}_4\text{-N}$ concentrations and $\text{NO}_3\text{-N}$ concentrations in soil ($k_{\text{Org-N}} = 100000.0$).

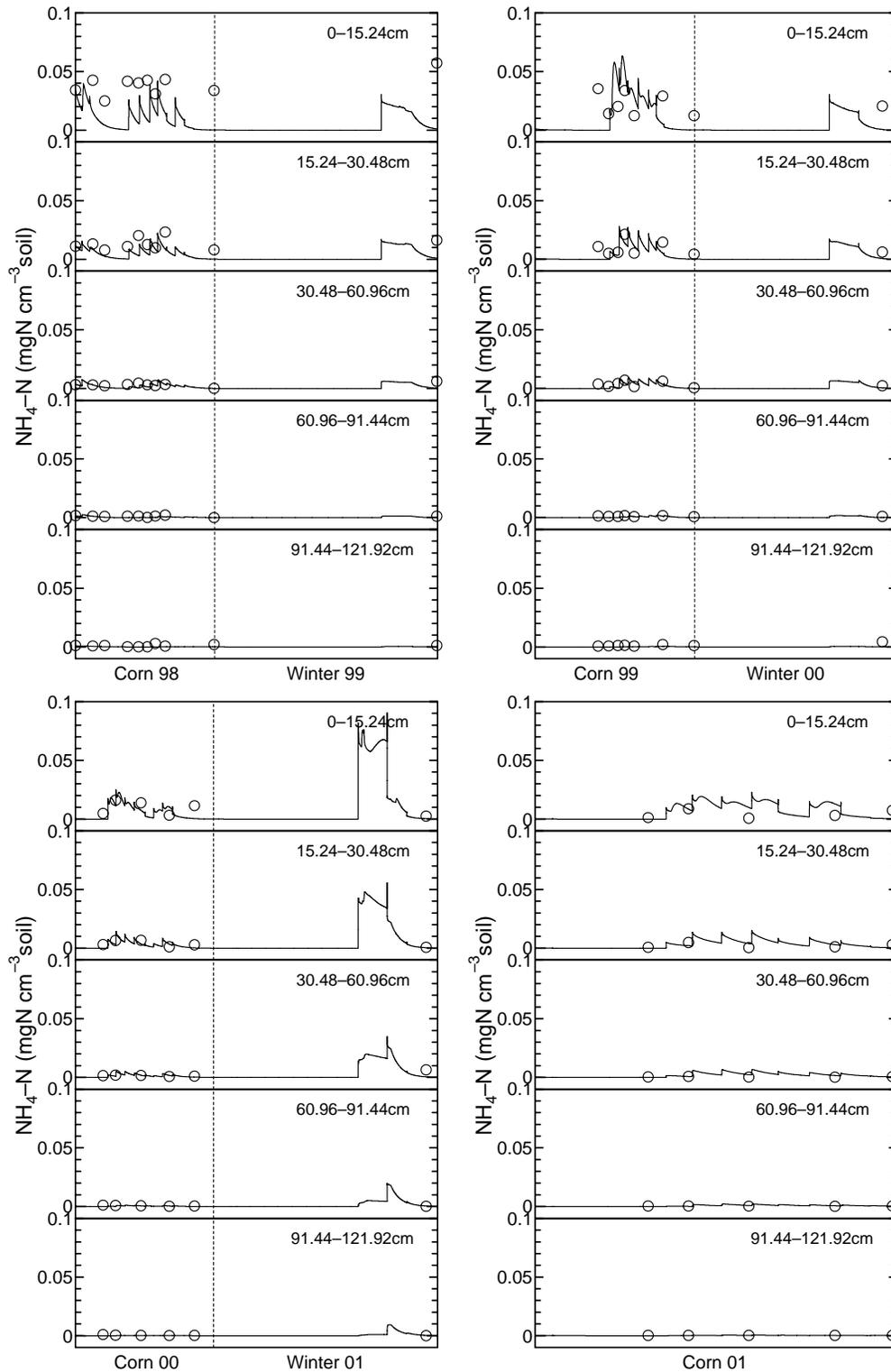


Figure 13: Measured and calculated temporal changes in total $\text{NH}_4\text{-N}$ in soil profiles for the case of the best nitrogen parameters set which produces the minimum root mean square error of soil nitrogen concentration.

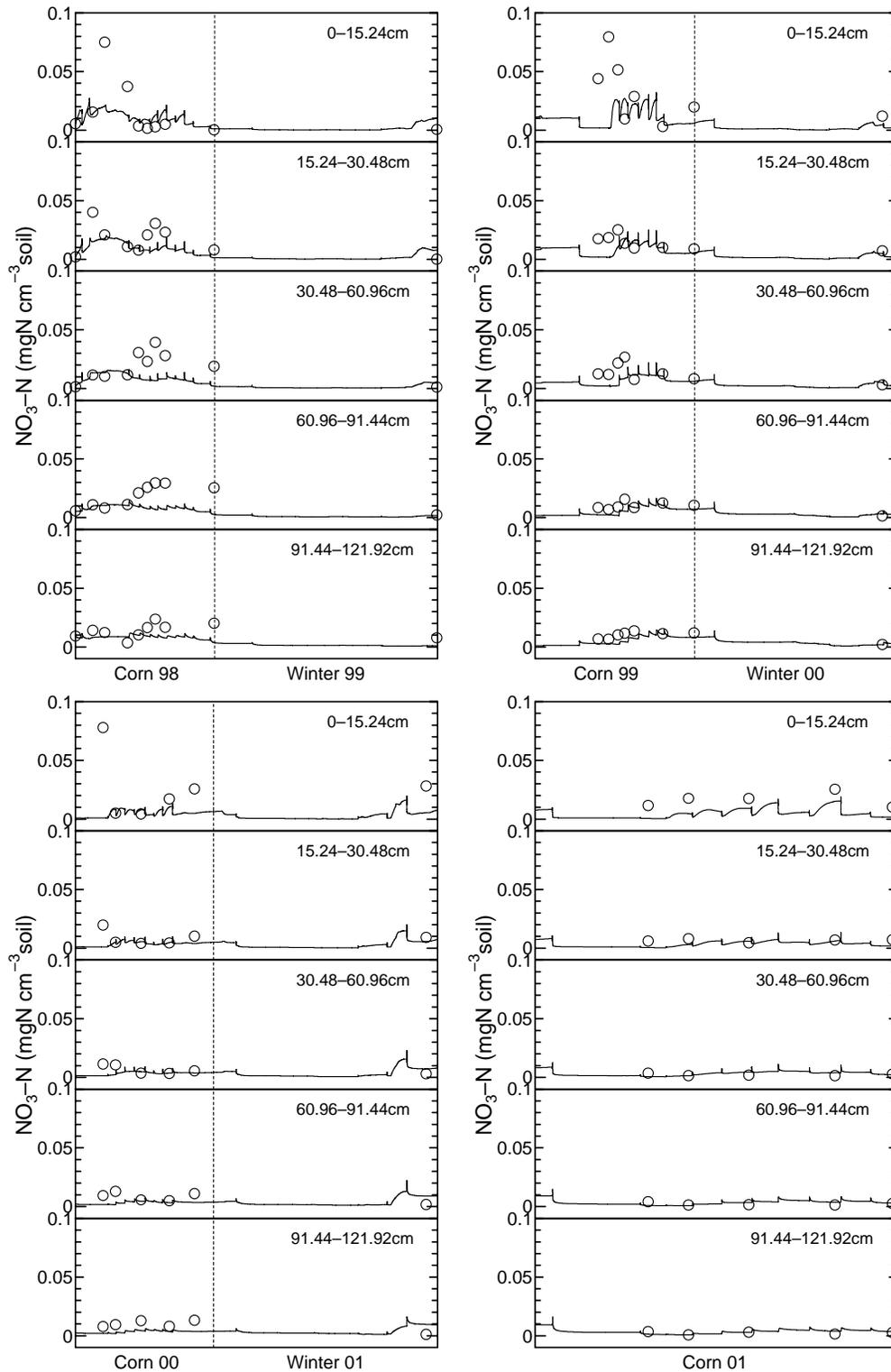


Figure 14: Measured and calculated temporal changes in $\text{NO}_3\text{-N}$ in soil profiles for the case of the best nitrogen parameters set which produces the minimum root mean square error of soil nitrogen concentration.

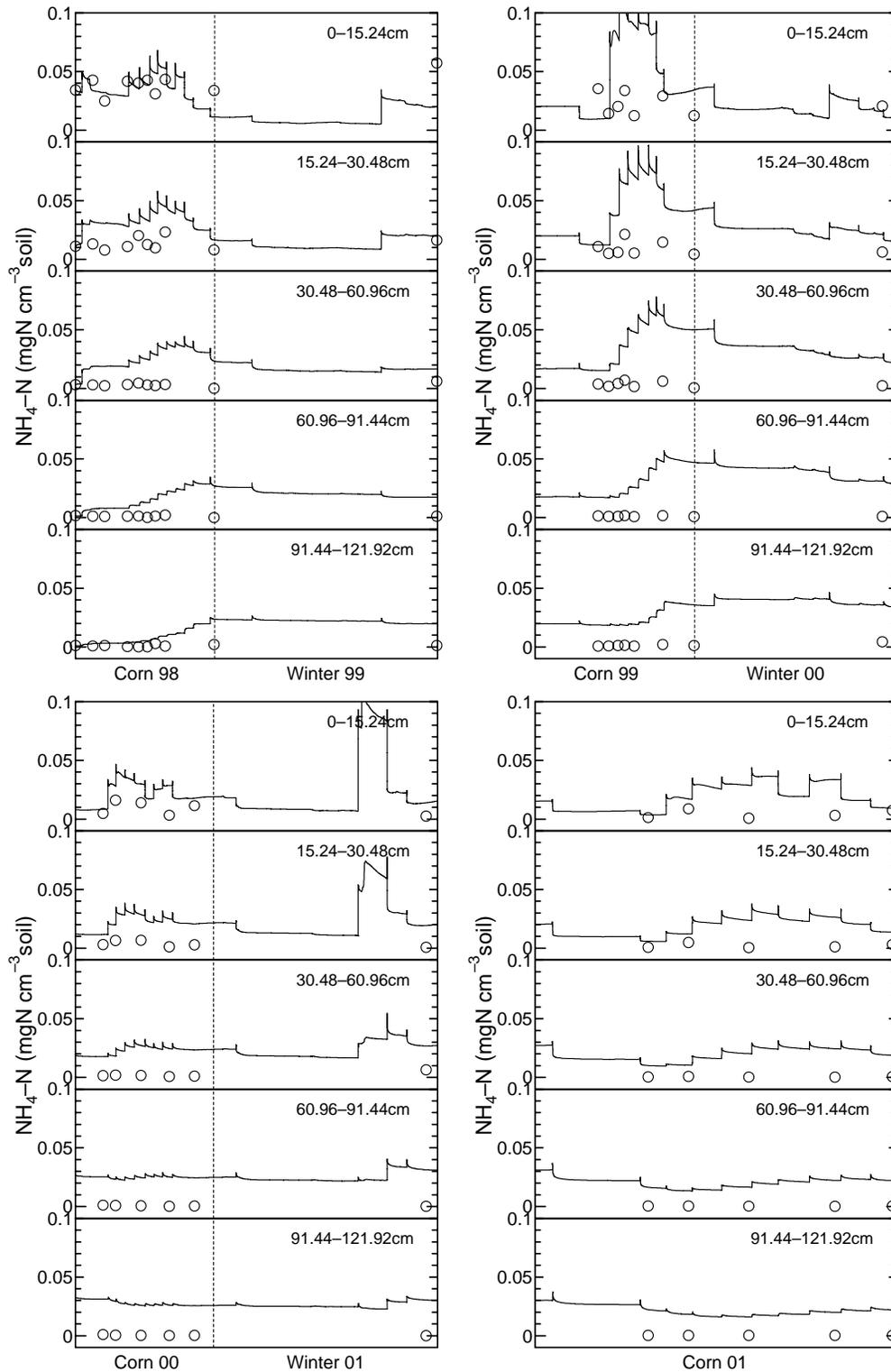


Figure 15: Measured and calculated temporal changes in total NH_4-N in soil profiles for the case of the worst nitrogen parameters set which produces the maximum root mean square error of soil nitrogen concentration.

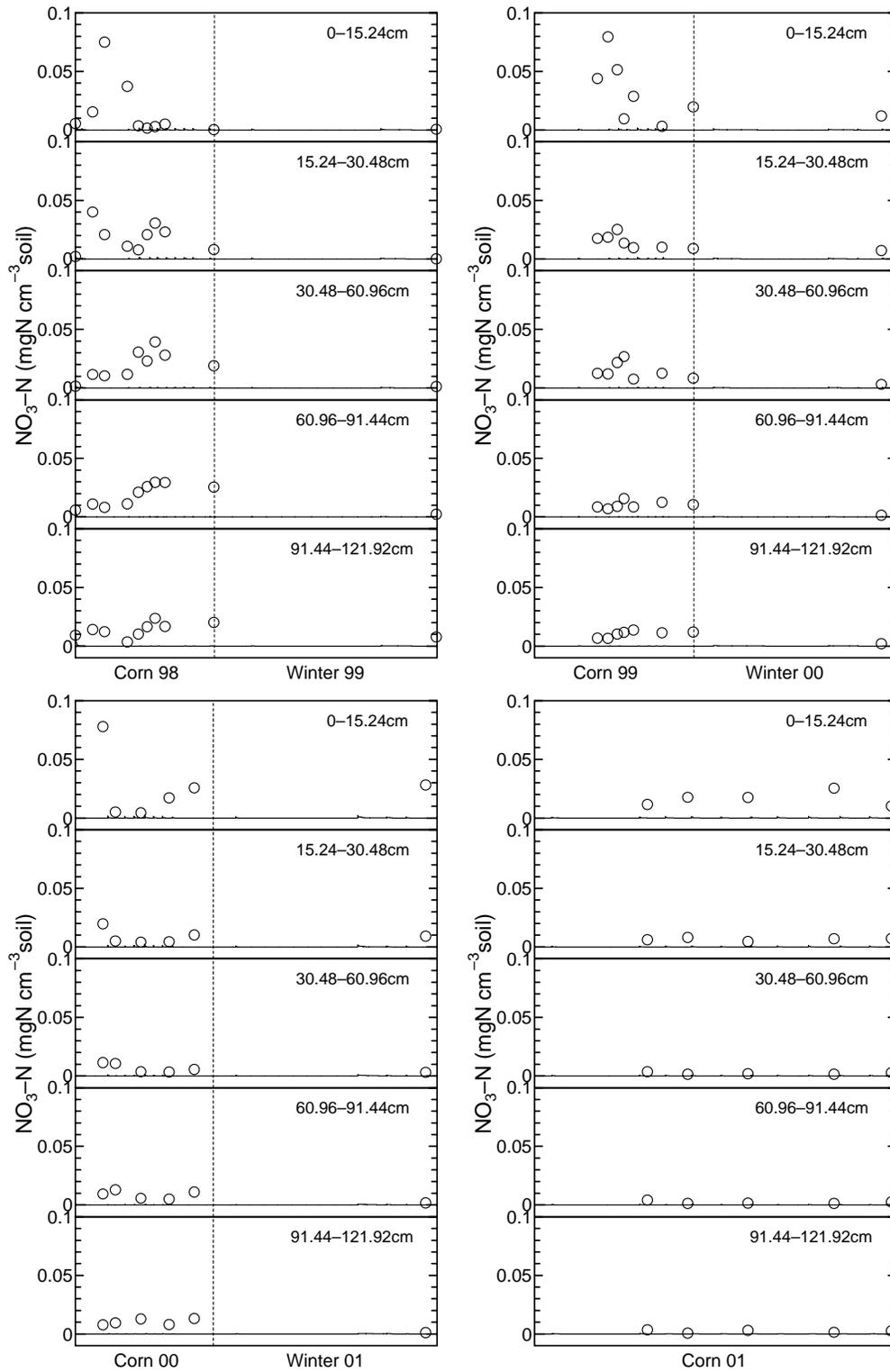


Figure 16: Measured and calculated temporal changes in $\text{NO}_3\text{-N}$ in soil profiles for the case of the worst nitrogen parameters set which produces the maximum root mean square error of soil nitrogen concentration.

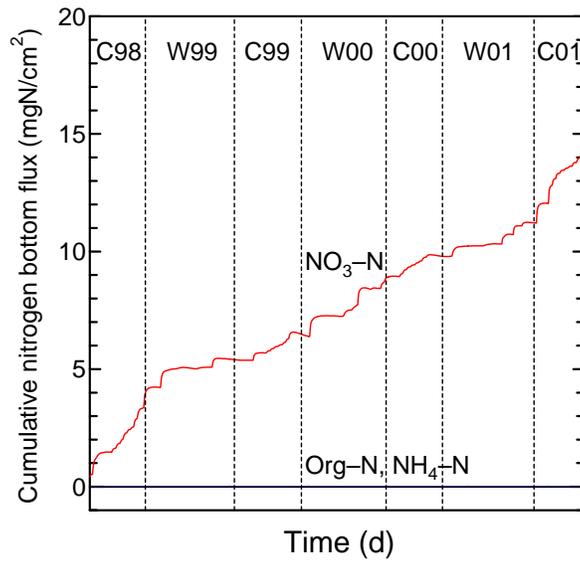


Figure 17: Cumulative *Org-N*, *NH₄-N* and *NO₃-N* fluxes at the bottom boundary (3m depth).

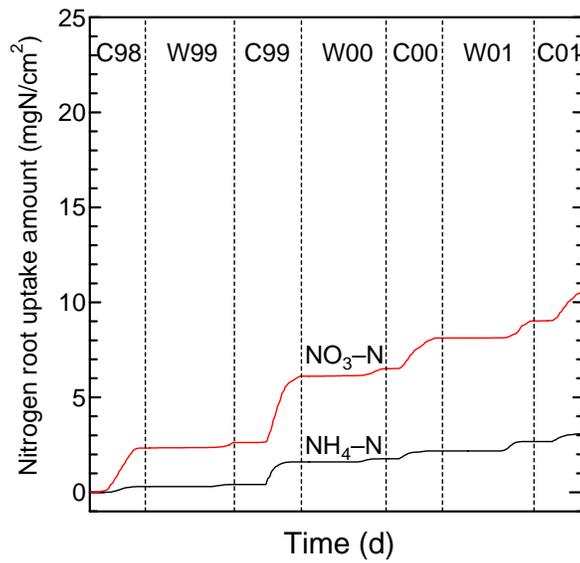


Figure 18: Cumulative root uptake amounts of *NH₄-N* and *NO₃-N*.

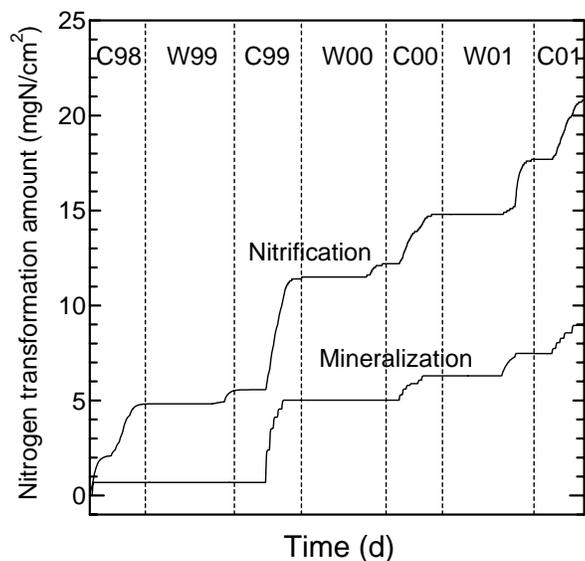


Figure 19: Cumulative mineralization and nitrification amounts.

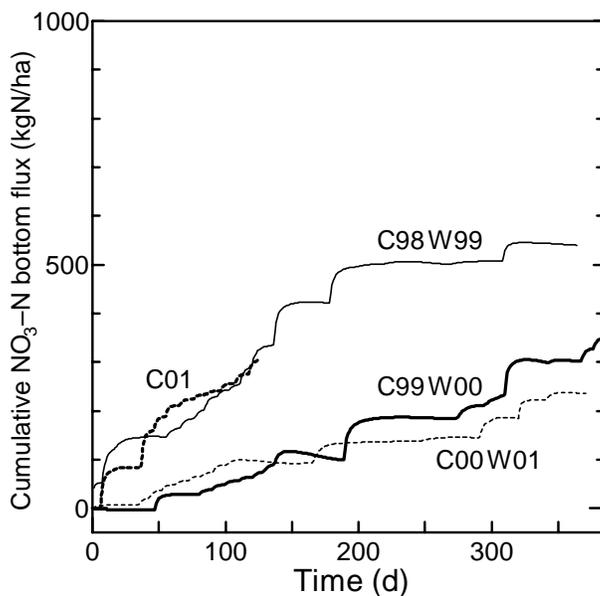


Figure 20: Cumulative $\text{NO}_3\text{-N}$ flux at the bottom boundary for each crop year. $\text{NO}_3\text{-N}$ loads into groundwater for Corn 99, Winter 00, Corn 00, and Winter 01 were smaller than for Corn 98 and Winter 99. Targeted management of manure was effective for controlling the nitrate groundwater contamination.

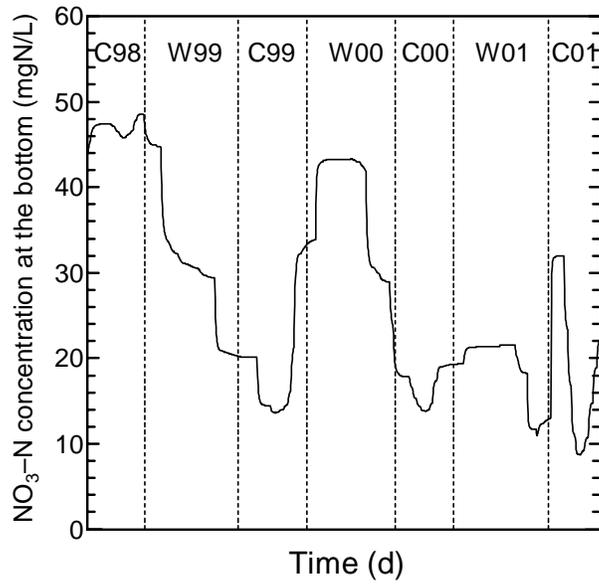


Figure 21: *NO₃-N concentration at the bottom (3m depth) for each crop year.*

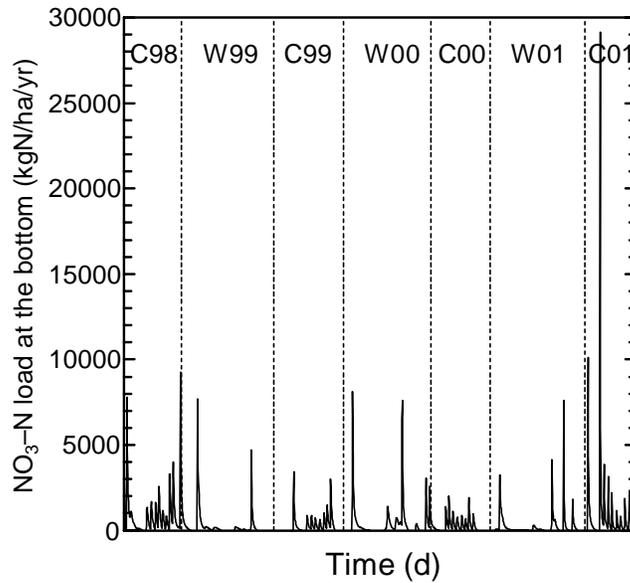


Figure 22: *NO₃-N load at the bottom (3m depth) into groundwater for each crop year. High NO₃-N load for Corn 01 is due to high irrigation water amount (Fig.4).*

Appendix I: California Biosolids Mineralization Research

PRESENTED AT CALIFORNIA CALIFORNIA PLANT AND SOIL CONFERENCE, 1998

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I.1 Introduction

The Clean Water Act charges the United States Environmental Protection Agency (USEPA) with regulating the disposal of sewage sludges generated during wastewater processing. These sludges are formed by concentrating the inert and organic solids that are either collected with the wastewater or generated during treatment. To help communities dispose of them, the USEPA actively promotes land application of sludges as an economical alternative to landfilling or incineration. Because sludges contain large amounts of the nitrogen, phosphorus, and trace elements needed by growing plants, they are promoted as fertilizers. They also can improve soils as organic amendments in the same way as animal manures ("Standards" 1993). To encourage public acceptance of land application, the USEPA and the wastewater treatment industry often refer to sewage sludges as *biosolids* ("Biosolids" 1994, "Plain" 1994, Sorber 1994).

Due to their origin in wastewater, biosolids contain a number of pollutants that require special attention. Biosolids contain pathogens, heavy metals, and toxic organic chemicals, along with nutrients that, if over-applied, can harm both people and the environment ("Standards" 1993, "Process" 1995, Committee 1996, Crohn 1996, Harrison *et al.* 1997). In general, the relative presence of pathogens, metals, and toxic organics will determine if a product may be put to a particular use, such as forage fertilization or backyard gardening, while the quantity actually applied in a given year or growing season is constrained by the nutrient content, or fertilizer value, of the biosolids.

Because California depends heavily on groundwater, nitrogen limits most land application rates. Over-fertilization with nitrogen often pollutes groundwater with nitrate (Patrick *et al.* 1987, Follett *et al.* 1991). Biosolids contain one to ten percent nitrogen on (a dry weight basis) the majority of which is in organic form ("Process" 1995). The balance consists of inorganic nitrogen is in the form of ammonium (NH_4^+), much of which can volatilize to the atmosphere during the application process, or nitrate (NO_3^-), which leaches readily and can also be lost to the atmosphere through denitrification ("Process" 1995, Crohn 1996). Only inorganic forms are immediately plant-available, but some of the organic nitrogen is converted each year to ammonium in a process called *mineralization*. Designers add the expected mineralized nitrogen to the inorganic nitrogen contained in the incorporated biosolids to determine application rates appropriate for specific crops. Little information is available about the rate at which biosolids organic nitrogen is miner-

alized in California soils, however. To reliably meet the “agronomic rate” requirement of 40 CFR Part 503, more research will be needed.

I.2 Mineralization and Related Processes

The organic fraction of biosolids nitrogen refers to that portion that is bound up with carbon compounds. Because plants cannot take it up, organic nitrogen is often referred to as *immobilized* nitrogen. To be used by plants, organic must be converted, or *mineralized*, to ammonium ions (NH_4^+) by soil microbes. Under aerobic, or oxygen rich, conditions, other bacteria may convert the ammonium to nitrate (NO_3^-), a process referred to as *nitrification*. Both nitrate and ammonium are plant available.

During mineralization, biosolids are used as food by bacteria and other microorganisms. Soil microbes use biosolids carbon as an energy source while biosolids nitrogen serves primarily as a nutrient for building proteins in new and growing cells. As carbon is used for energy, it is converted to CO_2 and released from the soil as a gas. This steadily depletes carbon from the soil. As carbon decomposes, any associated organic nitrogen is mineralized to ammonium. Different forms of carbon are metabolized at different rates, however. Carbohydrates and proteins are decomposed readily while more recalcitrant forms, such as lignin and cellulose, break down more slowly (Terry *et al.* 1979, Boyle 1990, Lerch *et al.* 1992).

Environmental conditions also affect mineralization rates which can vary considerably. The most important factors are biosolids type (Epstein *et al.* 1978, Parker and Sommers 1983, Garau *et al.* 1986, “Process” 1995), soil temperature (Terry *et al.* 1981), soil type (Tester *et al.* 1977, Magdoff and Amadon 1980, Garau *et al.* 1986, Dendooven *et al.* 1995), and moisture (Stanford and Epstein 1974). Other factors, such as application rate and soil pH are generally less important (Tester *et al.* 1977). Warm well-drained soils promote rapid mineralization rates. Such soils are common in California.

The USEPA has published a table of typical mineralization rates for biosolids (Table 1). Values found in Table 1 were loosely developed from a series of sixteen week laboratory incubations (Sommers *et al.* 1981), rather than from field studies. Although Sommers *et al.* (1981) maintained conditions they considered optimal for mineralization, conditions favoring mineralization can persist well beyond sixteen weeks in many parts of California. Sommers *et al.* (1981) recommended annual mineralization rates of 40% for aerobically digested biosolids, 15% for anaerobically digested biosolids, and 8% for composts. They did not specifically develop second year recommendations. The USEPA reduces mineralization rates by half for each additional year after application (Table 1). No justification is given for this approach, although a precedent similar to this began in California.

Table I.1: USEPA Example Mineralization Rates (“Process” 1995)

Time Years after application	Biosolids type		
	Aerobically digested	Anaerobically digested	Composted
0-1	0.30	0.20	0.10
1-2	0.15	0.10	0.05
2-3	0.08	0.05	-
3-4	0.04	-	-

Pratt *et al.* (1973) invented the idea of the *decay series* to describe the decomposition of biosolids and manure in California soils. They predicted that biosolids from a source they were familiar with would release 35% of its organic nitrogen during the first year after application, 10% of the remaining organic nitrogen during the next year and 5% over the third. An application containing 1000 kg organic nitrogen would therefore mineralize 350 kg, 65 kg, and 29 kg during the first, second, and third years after application. The USEPA included this fundamental approach, which is simple, reasonable, and intuitive, in Table 1. Note that the biosolids product considered by Pratt *et al.* (1973) was anaerobically digested and that their California decay series is therefore significantly higher than the USEPA’s (“Process” 1995). The particular values that make up the decay series are in doubt, however. Pratt *et al.* (1973) warn that their decay series “should not be considered appropriate for all municipal sludges.” They admit freely that their values were never tested in the field and recommended long-term trials to validate them. Although the USEPA has not subjected the decay series described in Table 1 to rigorous field testing either, these values for the basis for many, and probably most, current designs.

Mathematical modeling presents a more conceptually coherent alternative to the decay series approach. Mathematical models attempt to summarize the many complex processes we call decay using tractable and measurable parameters. First-order models are most often used to predict decay. The simplest form of first order model lumps all organic nitrogen into a single compartment (Stanford and Smith 1972, Parker and Sommers 1983, Garau *et al.* 1986, Boyle and Paul, 1989, Federle *et al.* 1997),

$$N_m = N_o(1 - e^{-kt}), \quad (1)$$

where N_m is the nitrogen mineralized during year t (kg/ha/yr), N_o is the initial organic nitrogen mass (kg/ha), and k is the annual mineralization rate (yr^{-1}), a constant. Other investigators represent two compartments (Molina *et al.* 1980, Lindeman and Cardenas 1984, Lerch *et al.* 1992, Gilmour *et al.* 1996), one for a rapidly metabolized, or *labile*, fraction, and the other for more *recalcitrant* materials. Both compartments decompose as first order processes,

$$N_m = N_o S(1 - e^{-k_l t}) + N_o (1 - S)(1 - e^{-k_r t}), \quad (2)$$

where S is the biosolids labile fraction, and k_l and k_r are annual mineralization rates for the labile and recalcitrant fractions (yr^{-1}), respectively. Gilmour *et al.* (1996) favor a variation on (2) based on sequential, rather than simultaneous, decomposition of the labile and recalcitrant compartments, an approach similar to Crohn (1996).

I.3 Field Studies

Data from field studies are needed to develop parameter values suitable for California if (1), (2), or similar models are to be used to design application rates. Few such studies have appeared in the literature, however. Artiola and Pepper (1992) conducted a five-year study of land application to a sandy irrigated soil in Arizona. Laboratory tests suggested annual mineralization rates of 65% or greater. Accelerated mineralization was confirmed in the field where it was observed that applications failed to significantly increase the soil total nitrogen pool. Nitrate levels increased substantially, however, confirming that almost all of the applied biosolids had mineralized.

Barbarick *et al.* (1996) published a study of mineralization rates from biosolids applied to dryland wheat in Colorado. They applied 5 to 6 application during the 11 year study which was hampered since no record was made of the nitrogen present in the experimental plots before applications began. Mineralization rates varied greatly according to application rates and ranged from 13 to 67% during the first year. Experimental error and environmental variability account for much of the observed variability.

Chang *et al.* (1988) incorporated a biosolids compost and two anaerobically digested biosolids products into a sandy loam soil plots as well as a loam soil plots located at the University of California field station in Moreno Valley, California. Between 1975 and 1983, the investigators harvested three sorghum and eight barley crops. The study included control plots as well as the biosolids-amended plots. The uncomposted biosolids decomposed very rapidly (Decay series: 0.89, 0.30, 0.10, 0.05) while the compost mineralized more slowly (Decay series: 0.47, 0.20, 0.10, 0.05). The authors did not model decomposition as a first-order process.

I.4 Discussion

Chang *et al.* (1988) developed the only scientifically rigorous mineralization values available for designing land application rates field tested in California. The hot irrigated desert climate of Moreno Valley strongly favors mineralization. The decay series reported by Chang *et al.* (1988) probably overestimates decay in dryland or cooler climates. Additional studies would help to validate or refute the Moreno Valley numbers.

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APPENDIX J: Salt Balances of Fields Receiving Dairy Wastewater Applications

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Dairy wastewater typically is loaded with nutrients (i.e. N, P, and K) and is high in salinity. Since 1999, Marsha Campbell-Mathews, Thomas Harter, and Roland D. Meyer have been conducting field experiments in which they applied blended dairy wastewater for forage productions in the San Joaquin Valley, employing double cropping of summer silage corn and winter wheatgrass. During the course of this investigation, they accumulated considerable amounts of data on the chemical composition of wastewater stored in the dairy wastewater lagoons and they monitored the chemical properties of groundwater underneath the application sites. We chose three examples to illustrate the range and the typical compositions of dairy wastewater and the salt balances of fields receiving dairy wastewater applications (Table J-1). Typical chemical compositions of the irrigation water sources used for blending at the east and west sides of the San Joaquin Valley are summarized as follows, based on information provided by Blake Sanden (Farm Advisor, UCCE Kern County):

Source	EC	PH	Ca ²⁺ (meq l ⁻¹)	Mg ²⁺ (meq l ⁻¹)	Na ⁺ (meq l ⁻¹)	K ⁺ (meq l ⁻¹)	SO ₄ ²⁻ (meq l ⁻¹)	Cl ⁻ (meq l ⁻¹)	HCO ₃ ⁻ (meq l ⁻¹)
East	0.05	7.6	0.2	0.1	0	0.2	0.1	0.2	0.2
West	0.5	7.6	2	1	0	2	2	2	1

The water applications satisfied the irrigation requirements and delivered adequate amounts of nutrients for the growing crops. The blending ratios for the dairy wastewater and irrigation water were typically 1 to 10 to 1 to 20 depending on the N contents of the wastewater. In addition to the commonly expected dissolved mineral ions (i.e., Ca²⁺, Mg²⁺, Na⁺, K⁺, SO₄²⁻, Cl⁻, and HCO₃⁻), the dairy wastewaters frequently are high in NH₄⁺ ion and in organic acid anions. Upon land applications, the NH₄⁺ cation will be absorbed directly by plants or be oxidized to nitrate and then absorbed. The organic acid anions were not separately analyzed. Under routine water quality chemical analysis protocols, their presence is reflected by alkalinity of the water. The organic acid anions typically will be oxidized to form components of bicarbonate (CH₃COO⁻ + 2O₂ → CO₂ + H₂O + HCO₃⁻). The carbonate-bicarbonate species in the soil solution in turn are in equilibrium with CO₂ in the soil atmosphere. For salt balance and salinity assessment, these two components of the dissolved substances in dairy wastewater do not affect the final outcomes and may not need to be considered.

According to Campbell-Mathews, the annual double cropping of summer corn and winter wheat grass requires 250 and 150 lbs ac⁻¹ (280 and 168 kg ha⁻¹) of nitrogen inputs, respectively; receives on the average 12 inches (30.5 cm) of precipitation, 10 inches (25.4 cm) of winter irrigation, and 36 inches (91.4 cm) of summer irrigation; and has leaching fractions between 0.28 and 0.31 (Campbell-Mathews et al., undated). Typically, the K uptake by a 30-tons-per-acre silage corn crop would reach 250 lbs K₂O ac⁻¹ (280 kg ha⁻¹). When K is available, the luxury uptake by plants might consume an additional 100 lbs K₂O acre⁻¹ (112 kg ha⁻¹). Campbell-Mathews et al. (Undated) reported that over 75% of lagoon water samples contained more K than

plants might consume. Using the dairy wastewater lagoon at Redbun Dairy as an example, we estimated the salt leaching when the dairy wastewater was blended with irrigation water sources from the east and west sides of the San Joaquin Valley (Tables J-2 to J-5) in double cropping of summer corn and winter forage. In the computations, we assumed the total annual plant K uptake to be $500 \text{ kg K}_2\text{O ha}^{-1}$. This amount, along with corresponding anions in bicarbonate form, was deducted from the salt inputs prior to using WATSUIT (Oster and Rhoades, 1990; Rhoades et al., 1992). It was then compared with the salinity changes caused by using east and west side irrigation water sources alone (Tables J-6 to J-9). The TDS and the ionic compositions of the drainage water were compared with the chemical characteristics of groundwater samples collected at the Redbun dairy wastewater application site (Table J-10). Relatively, the K^+ content of the groundwater was significantly lower than that in the projected drainage water, and the Ca^{2+} and Mg^{2+} contents of the groundwater were significantly higher than those in the projected drainage water. Potassium ions are known to be specific-adsorbed in mineral lattices of the 2:1 layer silicate clay minerals such as micas (Fanning and Keramidas, 1977). Many soils in San Joaquin Valley exhibited K adsorption characteristics. It appeared that the K in the drainage water was specifically adsorbed by the mica clay minerals and substituted by Ca^{2+} and Mg^{2+} ions. If the K ion concentrations of the drainage water in Tables J-2 – J-5 are adjusted for the K^+ adsorption by clay minerals and replaced by Ca^{2+} ion, the resulting chemical compositions resemble those of the groundwater as shown in Table J-10.

The plant K uptake adjusted annual salt loading for the four scenarios is summarized in Table J-11. The use of dairy wastewater could increase the annual salt loading by 3000 to 3500 kg ha^{-1} . Lime precipitation does not appear to play a significant role in reducing the salt loads to ground water. The groundwater underneath the dairy wastewater application fields will also experience significant increases in hardness and alkalinity.

Table J-1: *Chemical composition of selected dairy wastewater in the San Joaquin Valley*

Dairy	Date	EC (dS m ⁻¹)	pH	Cation (meq l ⁻¹)					Anion (meq l ⁻¹)			TDS (mg l ⁻¹)
				Ca ²⁺	Mg ²⁺	Na ⁺	K ⁺	NH ₄ ⁺	SO ₄ ²⁻	Cl ⁻	Alkalinity	
TC20	6/29/00	1.34	7.0	3.9	2.6	11.9	0.6	1.2	1.1	3.4	10	780
TC11	6/27/00	6.02	7.5	6.9	7.6	12.8	17.5	37.8	0.8	14	63	2388
Redbun	7/11/00	9.86	7.2	10	11.9	15.2	24.8	21.7	0.8	14	-	-

Table J-2: *K uptake adjusted salinity of applied and drainage water of winter forage irrigation, wastewater from Redbun blended with east side water sources*

Water Source	EC (dS m ⁻¹)	pH	TDS (mg l ⁻¹)	Cation (meq l ⁻¹)				Anion (meq l ⁻¹)			
				Ca ²⁺	Mg ²⁺	Na ⁺	K ⁺	SO ₄ ²⁻	Cl ⁻	CO ₃ ²⁻	HCO ₃ ⁻
Applied	-	-	319	0.68	0.76	1.50	1.00	0.09	0.92	-	2.92
Drainage	1.03	7.13	804	2.27	2.53	3.33	2.44	0.3	3.10	0.29	6.99

Table J-3: *K uptake adjusted salinity of applied and drainage water of summer corn irrigation, wastewater from Redbun blended with east side water sources*

Water Source	EC (dS m ⁻¹)	pH	TDS (mg l ⁻¹)	Cation (meq l ⁻¹)				Anion (meq l ⁻¹)			
				Ca ²⁺	Mg ²⁺	Na ⁺	K ⁺	SO ₄ ²⁻	Cl ⁻	CO ₃ ²⁻	HCO ₃ ⁻
Applied	-	-	343	0.80	0.82	1.12	1.52	0.14	1.05	-	3.08
Drainage	1.07	7.13	826	2.67	2.73	3.78	1.96	0.47	3.50	0.19	6.94

Table J-4: *K uptake adjusted salinity of applied and drainage water of winter forage irrigation, wastewater from Redbun blended with west side water sources*

Water Source	EC (dS m ⁻¹)	pH	TDS (mg l ⁻¹)	Cation (meq l ⁻¹)				Anion (meq l ⁻¹)			
				Ca ²⁺	Mg ²⁺	Na ⁺	K ⁺	SO ₄ ²⁻	Cl ⁻	CO ₃ ²⁻	HCO ₃ ⁻
Applied	-	-	434	1.39	1.11	1.71	1.50	0.84	1.63	-	3.23
Drainage	1.61	7.17	1182	4.63	3.70	5.69	2.44	2.80	5.43	0.34	7.89

Table J-5: *K uptake adjusted salinity of applied and drainage water of summer corn irrigation, wastewater from Redbun blended with west side water sources*

Water Source	EC (dS m ⁻¹)	pH	TDS (mg l ⁻¹)	Cation (meq l ⁻¹)				Anion (meq l ⁻¹)			
				Ca ²⁺	Mg ²⁺	Na ⁺	K ⁺	SO ₄ ²⁻	Cl ⁻	CO ₃ ²⁻	HCO ₃ ⁻
Applied	-	-	618	2.49	1.67	2.81	1.52	1.93	2.74	-	3.83
Drainage	2.33	7.16	1629	7.01	5.57	9.37	1.96	6.43	9.10	0.43	6.43

Table J-6: *Salinity of applied and drainage water of winter forage irrigation with east side water sources*

Water Source	EC (dS m ⁻¹)	pH	TDS (mg l ⁻¹)	Cation (meq l ⁻¹)				Anion (meq l ⁻¹)			
				Ca ²⁺	Mg ²⁺	Na ⁺	K ⁺	SO ₄ ²⁻	Cl ⁻	CO ₃ ²⁻	HCO ₃ ⁻
Applied	-	-	15	0.091	0.045	0.091	-	0.045	0.091	-	0.091
Drainage	0.16	5.82	55	0.30	0.15	0.30	-	0.15	0.30	-	0.30

Table J-7: Salinity of applied and drainage water of summer corn irrigation with east side water sources

Water Source	EC (dS m ⁻¹)	pH	TDS (mg l ⁻¹)	Cation (meq l ⁻¹)				Anion (meq l ⁻¹)			
				Ca ²⁺	Mg ²⁺	Na ⁺	K ⁺	SO ₄ ²⁻	Cl ⁻	CO ₃ ²⁻	HCO ₃ ⁻
Applied	-	-	34	0.2	0.1	0.2	-	0.1	0.2	-	0.2
Drainage	0.24	6.15	113	0.67	0.33	0.67	-	0.33	0.67	-	0.67

Table J-8: Salinity of applied and drainage water of winter forage irrigation with west side water sources

Water Source	EC (dS m ⁻¹)	pH	TDS (mg l ⁻¹)	Cation (meq l ⁻¹)				Anion (meq l ⁻¹)			
				Ca ²⁺	Mg ²⁺	Na ⁺	K ⁺	SO ₄ ²⁻	Cl ⁻	CO ₃ ²⁻	HCO ₃ ⁻
Applied	-	-	148	0.91	0.45	0.91	-	0.91	0.91	-	0.45
Drainage	0.75	6.47	494	3.03	1.50	3.03	-	3.03	3.03	0.01	1.51

Table J-9: Salinity of applied and drainage water of summer irrigation with west side water sources

Water Source	EC (dS m ⁻¹)	pH	TDS (mg l ⁻¹)	Cation (meq l ⁻¹)				Anion (meq l ⁻¹)			
				Ca ²⁺	Mg ²⁺	Na ⁺	K ⁺	SO ₄ ²⁻	Cl ⁻	CO ₃ ²⁻	HCO ₃ ⁻
Applied	-	-	326	2	1	2	-	2	2	-	1
Drainage	1.51	6.78	1085	6.67	3.33	6.67	-	6.67	6.67	0.08	3.25

Table J-10: *Chemical compositions of groundwater below Redbun Dairy (based on 22 samples collected from July 17, 2000 through June 5, 2001)*

Cation	Minimum	Maximum	Mean	Std. Dev.	Anion	Minimum	Maximum	Mean	Std. Dev.
Ca ²⁺	1.60	9.00	4.68	1.74	NO ₃ ⁻	0.79	4.99	2.80	0.91
Mg ²⁺	0.90	4.60	2.19	0.87	Cl ⁻	0.20	2.70	1.03	0.62
Na ⁺	1.00	4.30	2.06	0.86	HCO ₃ ⁻	2.05	9.90	3.84	2.25
K ⁺	0.02	1.47	0.24	0.43	SO ₄ ²⁻	0.23	0.70	0.48	0.13
NH ₄ ⁺	n.d.	0.006	0.003	0.001					

Table J-11: *K uptake adjusted salt loading of dairy wastewater application fields*

Scenario	Salt Input (kg ha ⁻¹)		Lime Precipitation (kg ha ⁻¹)	Annual Salt Loading (kg ha ⁻¹)
	Winter Forage	Summer Corn		
East Side Sources	86	310	0	404
Wastewater + East Side	1356	2284	0	3615
West Side Sources	828	2983	0	3794
Wastewater + West Side	2000	4792	0	6452

*Annual Summer Corn/Winter Forage Double Cropping with 250 and 150 lbs per acre of N inputs, respectively; annual water inputs are rainfall 12 inches (30.48 cm), winter irrigation 10 inches (25.4 cm), and summer irrigation 36 inches (91.44 cm); and leaching fraction is 0.3.