NITROGEN DYNAMICS ON PASTURE-BASED DAIRY FARMS IN GEORGIA

by

NATHANIEL P. EASON

(Under the Direction of Miguel Cabrera)

ABSTRACT

Information on nitrogen inputs and outputs for southeastern US pasture-based dairies is very limited. The first objective of this research was to estimate N losses via ammonia volatilization, nitrous oxide emissions, and nitrate leaching on two pasture-based dairies located in the Coastal Plain of Georgia (U.S.). The second objective was to create a one-year N balance for the two dairies. Nitrogen inputs were obtained from farm records. Nitrogen outputs included milk export, ammonia volatilization, nitrous oxide emission, and nitrate leaching. A micrometeorological passive flux method measured ammonia. Nitrous oxide emissions were measured using closed chambers in the field. Cup lysimeters were installed at 1-m depth to measure NO_3 leaching. Total N inputs were similar for both farms and so was N exported in milk when expressed as percentage of N inputs. However, ammonia and nitrate losses were very different between farms because of differences in N management practices.

INDEX WORDS: Grass-based dairies, N balance, Environmental impact, Dairy, Nitrate, Ammonia

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CHAPTER 1

INTRODUCTION

Purpose of the Study

Animal confinement systems are the primary type of dairy farm in Southeastern USA. Because of increasing grain and energy costs, these farms are becoming costly to operate. Dairies that use management-intensive grazing provide an alternative to confinement systems in Georgia and the Southeast. Pasture-based dairies generally have lower operating costs, better animal health, less environmental impact, and better energy efficiency than animal confinement dairies (Soder and Rotz, 2001). Nitrogen fertilization is one of the most important management practices in pasture-based dairies. Thus, efficient use of nitrogen is crucial for the sustainability of these dairies. Nitrogen cycles between the soil, water, and atmosphere components of the grassland system at rates which are poorly defined (Rotz et al., 2005). To understand the importance of N application on pasture-based dairies, it is necessary to look into the dynamics and processes associated with N fertilizer. Once N fertilizer is applied to a pasture, it may be leached below the rooting zone in the form of nitrate, or it may be lost as ammonia gas and/or nitrous oxide. Vaio et al. (2008) showed that up to 50% of the urea-N applied to a tall fescue pasture could potentially be lost as ammonia. Korevaar (1992) found the same percentage could be lost via nitrate leaching. Ledgard et al. (1999) discussed the potential for 16 to 30% of the added N to be immobilized in the soil as organic N one year after application.

Stricter legislative, economic, and public pressures demand that N use is optimized and that losses do not exceed thresholds, allowing both environmental and economic goals to be attained (Jarvis, 1993). By monitoring the N inputs and outputs on current operational pasture-based dairies in the Coastal Plain of Georgia, a better understanding of the N use efficiency. As a result, improved management practices can be developed/recommended that will make these operations more environmentally and economically sustainable.

Objectives

The first objective of this research was to estimate N losses via ammonia volatilization, nitrous oxide emissions, and nitrate leaching on two pasture-based dairies located in the Coastal Plain of Georgia. The second objective was to create one-year N balances for both farms based upon measured N losses and N inputs from farm records.

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CHAPTER 2

LITERATURE REVIEW

Confinement vs. Pasture-based Dairies

NRCS (2007) defines grazing-based dairy production systems as land use and feed management systems that optimize the intake of forages directly by grazing cows. It also defines confinement-based dairy production systems as land use and feed management systems that optimize milk production with confined cows consuming harvested forages. Gaining knowledge of energy, labor, and cost efficiencies within these farms will help provide current or future dairy farmers with valuable information to increase their profitability and optimize production on their farms. In addition, stricter environmental regulations regarding nutrient losses, waste handling, and water management are forcing dairies managers to implement environmental practices on the farm.

Confinement systems are capital-intensive, labor-intensive, and resource-intensive. These high costs, along with unstable or low milk prices, drastically increase risk and reduce long-term profitability for the farm operation (Soder and Rotz, 2001). Hoof and leg problems, acidosis, udder sores, mastitis, and general animal stress are also commonplace for the cows in confinement systems (NRCS, 2007). Grazing systems generally have reduced facility and labor costs, healthier cows with longer productive lives, and potentially a better quality of life for the dairy farmer (NRCS, 2007; Soder and Rotz, 2001; Weil and Gilker, NA).

Why Georgia?

Pasture-based dairy farms have been the primary form of dairy farming in many Western European countries as well as in New Zealand and Australia for decades. Now, more attention has been directed toward introducing these grazing systems into the southeastern United States, mainly the Coastal Plain of Georgia. So why choose Georgia? To begin with, the southeast is a milk-deficient region. In fact, the southeastern United States produces less than 8% of the national milk supply (Anon., 2004). The population in the Southeast continues to grow, with Georgia having the 4^{th} highest growth rate in the U.S. (Newberry, 2008). Therefore, an increase in the population demands for more milk producers. Ironically, of the milk produced in Georgia, 51% is exported, and of that, 48% is exported to Florida (Newberry, 2008); therefore, most of Georgia is dependent upon other states for its milk.

Additional attractions for implementing pasture-based dairies in Georgia are its climate and forage species that allow for a year-round grazing season (Ball et al., 2002). The sub-tropical climate allows optimum growth rates for forages such as bermudagrass and winter annual grasses. The Coastal Plain region of Georgia has an adequate supply of water which aids in the irrigation needed for the success of these farms. Topsoils within this region are generally described as highly weathered sandy loams (with varying depths) with low water-holding capacity, underlined by kaolinitic clay subsoils that are acidic, limit root growth, and have low cation exchange capacity. The low clay content of the upper 12 to 24 cm of soil profile tolerates traffic from cows and is well drained. As a result, cows graze on drained soils (as opposed to muddy pastures), which leads to increased herd health.

Nitrogen Balance

Little is known about the transport and losses of N on pasture-based dairies in Georgia's Coastal Plain. This lack of information is problematic given the rate at which the pasture-based dairy industry is being introduced in the Southeast. Little research has been done to determine the extent rotationally-grazed, irrigated pastureland on Coastal Plain soils effects the N cycle in these soils. Much research has been done in Northwestern Europe, Netherlands, New Zealand, and Australia on N balances and N losses on pastureland, however, extrapolation of this research is inappropriate to the Southeast because of differences in management practices, soil types, and climate. Transformations of N into different components as it cycles through the farm may cause large losses to the atmosphere, ground-, and surface-water (Rotz et al., 2005). Because N fertilizer is one of the largest costs for these farms, and little is known about the N dynamics in this region under such conditions, the purpose of this study is to obtain information on how N cycles under these conditions. For this reason, a whole farm N balance is a necessary first step because it is the scale at which nutrient management decisions are made. A whole farm approach also provides insight into how the N dynamics may affect other components within the system. For the N balance used in this study, our N inputs are N fertilizer and supplemental feed. The outputs are nitrous oxide (N_2O) emissions, ammonia (NH_3) volatilization, nitrate (NO_3) leaching, and milk N export. The following section provides a closer examination of these outputs.

Nitrous Oxide Loss

Nitrous oxide (N_2O) emission is thought to be a major contributor to global climate change. It is estimated that N_2O molecule has an impact equivalent to that of 300 CO_2 molecules (Turner et al., 2008). Nitrous oxide is produced by microbial processes that reduce NO_3 to N_2 (denitrification) and by the oxidation of NH_4 to NO_3 (nitrification) (Eckard et al., 2003). These processes are affected by moisture conditions, water-filled pore space (WFPS), oxygen status, soil temperature, N availability, organic matter, topography, and pH (Katayanagi et al., 2008; Turner et al., 2008). The complexities and interactions between biological, chemical, and physical properties of the soil lead to highly variable N_2O emissions both spatially and seasonally (Liebig et al., 2008; Parkin 1993; Senbayram et al., 2009; Carter 2007; Turner et al., 2008). Turner et al. (2008) performed a correlation matrix on the interactions between various physical and chemical soil properties, and found that the amount of NO_3 present in the soil was the most significant predictor of N_2O emissions. Liebig et al. (2008) also found a strong correlation between N_2O and WFPS in a study with cattle urine. Optimal N_2O fluxes seem to occur when the soil is between 60 and 80% WFPS, but emissions can still occur up to field capacity (Velthof and Oenema, 1995). Both, animal excreta deposited on a pasture and irrigation is expected to add to the variability (Turner et al., 2008). Results by Luo et al. (2008) showed that most N_2O fluxes reached a maximum at 1 to 14 days after grazing. In that study, N_2O emissions exhibited high seasonal variation, with rates decreasing in the following order: spring/early summer > late autumn/winter> summer/early autumn.

In grazed grassland systems, urea found in urine patches contributes to N_2O fluxes due to its N and C contents (Liebig et al., 2007). Oenema (1997) reported up to 3.8% on the N in urine is released as N_2O. Studies aimed at reducing GHGs from dairies have focused on modifying the animal's diet with a lower protein feed (Liebig et al., 2008; Luo et al., 2008). Urea in N fertilizers also causes large amounts of N_2O emissions. Ledgard et al. (1999) reported that gaseous losses by denitrification and volatilization increased 5-fold with the application of Urea fertilizer. In that study, N_2O loss was 5 kg N $ha^{-1}yr^{-1}$ for control, and 10 kg N $ha^{-1}yr^{-1}$ for ammonium nitrate and urea treatments. Extrapolation of data from other research to any location is not recommended because of the complexity of interactions of various biological, chemical, and physical properties that affect N_2O fluxes.

Ammonia Loss

A major concern on these pasture-based dairies is the loss of N via ammonia volatilization to the atmosphere. Bussink (1992) states there are limited data on the losses of NH_3 on grazed swards. The primary source of NH_3 losses on grazed pastures is thought to be urine deposited by cattle. Urine contains urea, which is hydrolyzed by the enzyme urease to produce ammoniacal N. The extent to which NH_3 volatilization occurs is dependent upon environmental factors and possibly grassland management. Ammonia losses to the atmosphere are important to manage because they may affect aerosol chemistry and acid deposition (Bussink, 1992). Deposition of NH_3 from the atmosphere could potentially lead to N loading to lakes, acidify soils of low buffering capacity, and damage sensitive crops (Gay and Knowlton, 2005; Pearson and Stewart, 1993; ApSimon et al., 1987; Van Breemen and Van Dijk, 1988). Recent studies in Europe show that up to 90% of the total NH_3 burden in the atmosphere originated from agriculture, with livestock wastes as its major source (Bussink and Oenema, 1998). While much is known about the effects NH_3 has on the atmosphere (Bussink, 1992), little is known about losses at the farm level. Gaining quantitative insight of the various sources of NH_3 volatilization and the factors that govern these losses is necessary to introduce cost-effective management practices. A whole-farm approach can consider the various components of NH_3 loss so that practices can be put into effect that minimizes this source of N loss.

Nitrate Leaching Losses

Loss of N in leaching of NO_3 through the soil profile raises many environmental concerns. Nitrate becomes a concern in groundwater when concentrations exceed USEPA's drinking water standard of 10 mg N L^{-1}. The probability of NO_3 leaching depends on climate, soil, and plant characteristics, with the highest losses in humid climates or irrigated systems, on coarse-textured soils or soils with artificial drainage, and under plants with short root systems, such as perennial ryegrass and white clover (Rotz et al., 2005). Various studies have shown that NO_3 leaching under grazed grasslands has been significantly higher than under cut grasslands (Hack-ten Broeke et al., 1996). This increase is because of the cattle excreta patches deposited on the field while animals graze. Roughly 80% of N consumed by grazing dairy cattle in feed and forages is deposited back into the pasture in the form of urine and feces (Whitehead, 1995). Dung patches have little impact on NO_3 leaching because 65 to 80% of the N excreted is contained in urine. It is estimated that one urine patch can represent a localized application ranging from 400 to 1200 kg N ha^{-1} (Hack-ten Broeke et al., 1996). However, a study done by White et al. (2001) showed that excreta from cattle under grazing conditions were relatively evenly distributed about the field after several grazing rotations. There have also been leaching studies considering dairy effluent application rates to various forages. Woodard et al. (2002) found that a Bermudagrass-annual ryegrass system compared to a corn-forage sorghum-rye system (where all three crops were annuals) was 20% more effective in N uptake and greatly reduced NO_3 leaching below the primary rooting zone. Despite the effective use of N by perennial grasses and annuals, NO_3 leaching can still pose environmental hazards and steps should be taken to minimize this source of N loss.

Another variable affecting NO_3 leaching on the farm is the use of supplemental irrigation. Irrigation is necessary on intensively managed pasture-based dairies to ensure an adequate forage supply for the animals. Little is known about how irrigated pastureland affects N leaching in Coastal Plain soils in temperate climates of the southeast region of the U.S. A study conducted in Denmark determined that a change from 25 to 15 mm per irrigation on sandy drier soils resulted in higher irrigation efficiencies and lowered annual water use from supplemental irrigation (Hack-ten Broeke, 2001). This study revealed that various irrigation strategies had no significant effect on NO_3 concentrations in the leachate for drier soils. Further research in site-specific locations would be necessary to understand the relationship between irrigation management and nitrate losses.

Milk Export

Nitrogen ouput in milk export is an important component of a farm-gate balance. . Ledgard et al. (1999) constructed a farm-gate N balance on clover/grass pastures grazed by dairy cows and assessed the influence of N fertilizer applications. They showed that fertilizer N inputs resulted in a minor increase in the N removed in milk relative to the amounts of N applied. Many other studies conducting N balances grouped the milk exported with the meat that was removed , labeling it as “milk and meat products” (e.g., Ledgard, 2001; Kristensen, 2004). As a result, it is difficult to determine what percentage of N was actually removed in milk (ADD figures on % removed in milk).

Modeling N flows

Some studies have used models models to predict N losses in intensive grazing systems. These models usually overestimate or underestimate N losses because of the pure complexity of how N cycles within these systems (e.g., McGechan and Topp, 2004; Watson and Atkinson, 1999). Since models are only as accurate as the calibration data they are based from, there is a need to study the N cycle in situations on pasture-based dairies in the Coastal Plain of Georgia. Gaining understanding of the N dynamics on these farms could potentially help improve the economic and environmental sustainability of this system.

Conclusions

There has been considerable research directed to the estimation of the N losses associated with dairy-slurry applications, leaching, volatilization, and N movement in dry-land grazing pastures, but these conditions are highly variable and site-specific. Monitoring the N dynamics on a grazing dairy in the Georgia Coastal Plain is needed. Research in other countries on pasture-based dairy farms indicates high losses of N to the environment are possible and suggests that monitoring of N flows on these farms is vital for their sustainability. Because of the possibility that pasture-based dairies will be an alternative practice to dairy farming systems in the Southeast USA, it is important to determine their environmental effects. For these reasons, this research is important to understand the environmental implications of pasture-based dairies within Georgia.

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CHAPTER 3

NITROGEN LOSSES IN TWO PASTURE-BASED DAIRIES ON GEORGIA'S COASTAL PLAIN (USA)

Eason, N., M.L. Cabrera, N. Hill, C. Mackowiak, and K. Roach. To be submitted to Nutrient Cycling in Agroecosystems

ABSTRACT

Management intensive pasture-based dairies are an alternative to traditional confinement dairy systems. Pasture-based dairies have lower operating costs, better animal health, and potentially less environmental impact. The objectives of this project were to estimate seasonal N losses through (NH_3) ammonia volatilization and nitrate (NO_3) leaching at two irrigated pasture-based dairies (Quitman and Wrens farms) located in the Coastal Plain of Georgia. Cup lysimeters were installed on each soil type to a depth of 1 m to monitor nitrate concentrations, which were used with water fluxes to estimate amounts of leached nitrate. Ammonia volatilization losses were measured with a micrometeorological technique that used passive ammonia samplers replaced every 30 to 60 days. Nitrate leached, expressed as % of applied N, amounted to 12.5 % at the Quitman farm and 0.3% at the Wrens farm. Larger losses at the Quitman farm were due to larger water fluxes coupled to larger nitrate concentrations at 1 m. Ammonia volatilization losses, expressed as % of applied N, amounted to 2% at the Quitman farm and 32% at the Wrens farm. The larger ammonia losses at the Wrens farm were attributed to surface applied urea compared to UAN applied through center pivot irrigation at the Quitman farm. Our results indicate that various N practices can affect the magnitude of NH_3 and NO_3 leaching losses, but overall indicate that these pasture-based dairies have losses similar to other grassland systems.

INTRODUCTION

Traditional, large-scale dairy farming in the USA has relied on confinement systems to rear and feed cows during lactation (NRCS, 2007). Most farms import feed from off-site locations, therefore importing nutrients in the grain onto the farm. Confinement farms may have limited land contiguous to the area where animal waste is collected, making it difficult to properly manage nutrients and often leading to on-site nutrient accumulation. Accumulation of nutrients such as nitrogen (N) and phosphorus in soil poses potential environmental problems because of the likelihood of contamination of surface runoff and ground waters (Osei et al., 2000). There is a trend towards increasing pasture use in USA dairies to accommodate the waste/nutrient distribution as well as providing economical benefits (Johnson, 2002). In a study with pasture grazing systems, White et al. (2001) found that manure was evenly distributed by the cows, which reduced manure handling and storage costs. Incorporating grazing into traditional dairies located in the southeastern USA is a logical management option because the region’s subtropical climate allows longer growing seasons for forages and nearly 12 months of grazing (Ball et al., 2002).

Nitrogen cycles between the soil, water, and atmospheric components of the grassland system at rates which are poorly defined (Rotz et al., 2005). These transformations are influenced by interactions among biological, chemical, and physical properties of the soil, climatic conditions, and grazing animals (Jarvis, 1993). Transformations of N in pasture ecosystems can cause large losses to the atmosphere as ammonia (NH_3) gas and to the groundwater as nitrate (NO_3) (Rotz et al., 2005).

Gaseous and leaching losses of N from pastures are not only economically important, but can pose a threat to the environment. Ammonia losses to the atmosphere may affect aerosol chemistry and acid deposition (Bussink, 1992). Recent studies in Europe show that up to 90% of the total NH_3 burden in the atmosphere originated from agriculture, with livestock wastes as its major source (Bussink and Oenema, 1998). Nitrate becomes an environmental concern in groundwater when concentrations exceed USEPA's drinking water standard of 10 mg N L^{-1}.

Most studies examining N losses in dairies examine systems in which dairy effluent is surface applied to pastures (e.g. Di et al., 1999; Williamson et al., 1998; Stevens and Logan, 1987; Misselbrook et al., 1996; Woodward et al., 2002). Hence most research on dairy N losses is not representative of a pasture-based system. Conversely, most research examining N dynamics in grassland systems have been conducted on dryland systems (e.g. Ball and Ryden, 1984; Jarvis and Ledgard, 2001; Di and Cameron, 2002; Ledgard et al., 1999; Bussink, 1992; Rotz et al., 2005) and therefore may not be applicable to irrigated pasture systems common to pasture-based dairies in the southeastern USA. Therefore, the objectives of this study were to: 1) estimate seasonal N losses via ammonia volatilization, and 2) estimate monthly leached NO_3 below a 1-m depth on two pasture-based dairies located in the Coastal Plain of Georgia.

MATERIALS AND METHODS

Farm Characteristics

Two commercial pasture-based dairy farms were chosen as experimental sites for this study. One farm (hereafter referred to as the Wrens farm) was located in east-central Georgia near the city of Wrens (Jefferson County). The second farm (hereafter referred to as the Quitman farm) was located in South Georgia, near the city of Quitman (Brooks County). The Wrens farm consisted of 124 ha and 500 cross-bred Jersey-Holstein milk cows, whereas the

Quitman farm had 114 ha and 450 Jersey-Holstein milk cows. Both farms grew forages under center pivot irrigation systems that had been divided into paddocks approximately 2.5 ha in size.

The dominant soil at the Wrens farm is an Orangeburg soil series (fine-loamy, Kaolinitic, thermic Typic Kandiudults) with class 0-3 and 3-7% slope classifications. The dominant soils at the Quitman farm included Tifton (fine-loam, siliceous, thermic plinthic, argillic Paleudults), Fuquay (loamy kaolinitic, thermic arenic plinthic Kandiudults), Stilson (loamy sand, Siliceous, subactive, thermic arenic plinthic Paleudults), and Alapaha series (loamy, siliceous, subactive, thermic arenic plinthic Paleaquults). The soils at both farms have loamy sand A_p horizons (approximately 25 cm deep) that transition to sandy clay loam B_t horizons.

Perennial pastures at the Wrens farm were planted to 'Tifton 85' bermudagrass (*Cynodon dactylon (*L.*) Pers.* – 16 ha) or 'Jesup Max Q' tall fescue (*Lolium arundinacea Schreb* – 23 ha). In addition, approximately 59 ha of prepared ground was planted to 'Tifleaf 3' pearl millet (*Pennisetum glaucum (*L.*) R. Br.*) in the spring and was later over-seeded with a mixture of 'Wrens Abruzzi' cereal rye (S*ecale cereal (*L.*) Salisb.*) and 'Feast 2' annual ryegrass (*Lolium multiflorum Lamarck*) in September. Bermudagrass pastures were also over-seeded with cereal rye. Tall fescue pastures were over-seeded with annual ryegrass in the fall of the year. The forage system at the Quitman dairy was 'Tifton 85' bermudagrass which was over-seeded with 'Florida 401' cereal rye and 'Big Boss' annual ryegrass in November.

Nitrogen Applications

Nitrogen applications in both farms were made throughout the study period as determined by the farm manager. Farm records were used to determine times and rates of these applications. At the Wrens farm, 78% of the N fertilizer was applied in the form of urea with about 22% of the

N supplied as poultry litter (134 kg N ha^{-1}). At the Quitman farm, N fertilizer was applied as urea ammonium nitrate (UAN) solution through the center pivot irrigation system.

Soil Analysis

Soils were sampled with a hydraulic probe in 25-cm increments to a depth of 100 cm in June 2008 at both farms. Six paddocks at the Wrens farm were sampled at eight randomly selected locations within each paddock. These paddocks represented the main soil type or pasture species present on the farm. Four paddocks were sampled in a similar manner at the Quitman farm, using one paddock for each soil type. Samples from the respective depths within a paddock were combined, air dried, and ground to pass a 2-mm screen. Soils were analyzed for cation exchange capacity (CEC) using an unbuffered salt extraction method (Sumner and Miller, 1996). Total carbon was determined in the top 25 cm of soil by dry combustion (Nelson and Sommers, 1996). A 1:1 ratio of soil and deionized water was used to measure pH with a Cornell pH meter (model 125, Corning Science Products, Medfield, MA). The pH buffering capacity was measured using a lime buffer capacity method (Kissel et al., 2007), and particle size analysis was determined with the pipette method (Gee and Or, 1996).

A second set of undisturbed soil core samples were collected from the different soil types on each farm to be analyzed for bulk density, hydraulic conductivity (Ks), gravimetric water content at field capacity (−0.03 MPa), and wilting point (−1.5 MPa). A soil water retention curve was also developed using Tempe pressure cells (Soil Moisture Equipment Corp., Santa Barbara, California). Saturated hydraulic conductivity was determined with a lab method in which a Mariotte bottle maintained a constant head of water ponded on the core, while flow rate measurements were recorded (Booltink and Bouma, 1996). Field capacity and wilting points were determined with a 1.5- MPa ceramic plate extractor (Soil Moisture Equipment Co., Santa Barbra, Ca) (Dane and Hopmans, 1996). To develop a soil water retention curve, soil cores were saturated in 0.01M $CaCl_2$, placed into a Tempe pressure cell, and exposed to 1.5, 2.9, 9, 18, 33, and 75 kPa of pressure (Flint and Flint, 1996). Retained water within the soil core was determined by weighing each apparatus daily until an equilibrium was reached at a particular pressure.

Environmental Monitoring

Environmental stations were placed in each soil type (Quitman farm) or forage type (‘Tift 85’ bermudagrass, ‘Jesup Max Q’ tall fescue, and ‘Feast 2’ annual ryegrass (Wrens farm) on the farms. Three environmental monitoring stations were installed on the Wrens farm and four on the Quitman farm. Because the soil at the Wrens farm (Orangeburg) was similar across the center pivot, the stations were placed in the different forage types. Each environmental station included a data-logger (Model CR206x, Campbell Scientific, Logan, UT), solar panel, rain gauge (TE525WS-L, Campbell Scientific, Logan, UT), temperature probe (Model 109, Campbell Scientific, Logan, UT), and three time domain reflectometry (TDR) probes (Model CS 625, Campbell Scientific, Logan, UT). The three TDR probes were placed 3 m into the paddock from an electric cross-fence line to measure water content at 0-25, 25-50, and 50-75 cm depths. The temperature probe was installed at a depth of 10 cm. The data-logger recorded temperature, rainfall/irrigation, and soil water content data every 15 min.

The TDR soil moisture probes were calibrated by collecting soil from the different depths at each station location. The soil was ground to pass through a 2-mm screen using a Wiley Mill, and packed in acrylic boxes (14.6 x 13.97 x 39.37 cm) to the bulk density measured in the field.

The TDR probes were inserted horizontally through holes on the side of the box so that they had 5 cm of soil above and below the probes. The soils were saturated with water and the boxes were placed in a greenhouse, where they were weighed twice daily for several days until a volumetric water content corresponding to -1.5 MPa (wilting point) was reached. Probe readings were recorded every hour to establish a mathematical relationship between mV reading and volumetric water content.

Ammonia Volatilization

Ammonia volatilization was monitored during 15 months at the Wrens Farm and during 13 months at the Quitman farm. Ammonia loss was determined using a passive flux method (Sommer et al., 1994). Passive flux samplers were installed on masts located on the periphery of the irrigation pivots (beyond the range of the irrigation water). The masts were installed at approximately 90° angles from each other, as close to the four cardinal directions as possible. Passive flux samplers were set at five heights (0.45, 0.75, 1.50, 2.25, and 3.00 m) on the masts (Leuning et al., 1985). Each passive flux sampler consisted of two glass tubes (each tube 0.7 i.d. by 10 cm long) connected by a piece of silicone tubing, with a nozzle connected to one of the tubes via another piece of silicone tubing. The nozzle consisted of a 2.3-cm-long (0.7-cm i.d.) glass tube with a stainless steel disk glued to it. The steel disk had a 1-mm-diameter hole at the center of the disk (Schjoerring et al., 1992). The inner surface of the tubes was coated with a 3% oxalic acid solution in acetone to trap ammonia. Two samplers were placed at each height. One open-ended tube in one sampler and one nozzle-ended tube in the other sampler faced toward the field to measure NH_3 volatilized from the land area under the center pivot. Similarly, one open-ended tube and one nozzle-ended tube faced away from the field to measure background NH_3.

Tubes were changed every 30 to 60 days and brought to the laboratory where they were extracted with 3 mL of deionized water to dissolve the ammonium oxalate formed as ammonia reacted with the oxalic acid. The extract was analyzed colorimetrically for NH_4 (Mulvaney, 1996) on an Alpkem auto-analyzer (RFA-300; Alpkem Corp., Clackamas, OR).

At each mast height, the horizontal flux of NH_3 from the center pivot through a plane perpendicular to the longitudinal axis of the sampler was calculated using Equation [1].

$$F_{hz,\,p} = [\,(C_1 + C_2\,)V] \,/\, (\pi\, r^2\, K\Delta\, t) \qquad [1]$$

where C1 and C2 are the concentrations of NH_4–N (μg N mL^{-1}) in the deionized water used to extract sorbed NH_4–N from the two tubes facing toward the field, V is the volume of deionized water use for extraction (3 mL), r is the radius of the hole in the stainless steel disk (0.5 mm), K is a correction factor (0.77), and Δt is the time during which the tubes were exposed (s). Similar calculations were performed with the NH_4 concentration in the two tubes facing away from the field to estimate the horizontal, background flux of NH_3 coming into the pivots ($F_{hz,b}$, μg N m^{-2} s^{-1}) through a plane perpendicular to the longitudinal axis of the sampler. The net horizontal flux of NH_3 derived from the field at each mast height was calculated by subtracting background fluxes from the fluxes coming from the center pivot.

The net vertical flux of NH_3 derived from the field (F_v, µg N m^{-2} s^{-1}) was estimated by integrating each net horizontal flux, taking into account the vertical distance corresponding to each mast height using Equation [2].

$$F_v = 1/R \sum_{h=1}^{h=5} (F_{hz,p} - F_{hz,b}) \Delta h \qquad [2]$$

where h represents each of the mast heights at which samplers were positioned, R is the radius from the center pivot, and Δh (m) is the vertical distance corresponding to each passive sampler (for any sampler, the distance between the point halfway to the sampler below and the sampler above it) (Schjoerring et al., 1992). Ammonia losses were expressed in kg N ha^{-1} per sampling period.

NO_3 Leaching

Five cup lysimeters were installed within 5 m of each of the environmental monitoring stations. A hydraulic probe was used to remove a soil core slightly longer than 1 m so that a lysimeter can be placed in the hole. Before placing the lysimeter in the hole, a clay/water slurry was poured in the bottom of the hole, to ensure contact between the soil and the ceramic cup. The lysimeter was inserted into the hole so that the cup was at a 1-m depth, and a 3:2 sand:kaolinite mixture was packed around the lysimeter tube to prevent vertical movement of water along the tube walls. Lysimeters were installed 3 m from one another along an electric fence-line, and were protected from animal trampling using an electric exclusionary fence. The lysimeters were placed under a -52 kPa vacuum to collect water samples bi-weekly. The water samples were analyzed colorimetrically for NO_3 on an Alpkem auto-analyzer (Mulvaney, 1996).

The results were averaged at each station on a monthly basis for a total period of 15 months at the Wrens farm and 13 months at the Quitman farm.

Soil water flow model (Hydrus 2-D)

A soil water flow model, Hydrus 2-D (Šimůnek et al., 2006), was used to determine the flux of water at a 1-m depth, which was the depth at which the lysimeters were collecting percolating water. Evapotranspiration, rainfall + irrigation, Ks, depth of horizons, texture analysis, bulk density, and the soil release curve parameters obtained with the Tempe cells were used in the Hydrus 2-D model to estimate the vertical water flux. The water flux leaving the bottom of the 1-m profile after each month was used to calculate the potential amount of nitrate leached by combining it with the nitrate concentration measured in the lysimeters.

RESULTS AND DISCUSSION

Soil Analysis

Analysis of soils on both farms showed characteristics typical of the Coastal Plain region (Perkins et al., 1986; NRCS, 2009) (Table 3.1). Soils in this region are acidic, with pH typically ranging from 4.6 to 6.0, becoming more acidic with depth. In all cases, except for the Stilson soil, the Ks decreased at the 25-50 cm depth. This is likely due to an increase of clay content beginning at this depth. The Fuquay soil had comparatively higher Ks values, which reflect the arenic nature of the epipedon. The 0-25 cm horizon of all soils was low in organic matter (<10 g C kg^{-1}) and low in CEC. Bulk densities also fell in range of well-drained soils found in Coastal Plain soils.

Fertilizer Application

Rates and total amounts of N fertilizer varied among farms. At the Wrens Farm, application rates ranged from 16 to 55 kg urea-N ha^{-1} mo^{-1} (Fig. 3.1), for a total application rate of 605 kg urea-N ha^{-1} (Table 3.2). At the Quitman Farm, applications ranged from 14 to 43 kg UAN-N ha^{-1} mo^{-1} (Fig. 3.1), with a total application rate of 320 kg UAN-N ha^{-1} (Table 3.2).

Ammonia Volatilization

In general, the amount of N lost through ammonia volatilization in each sampling period was greater at the Wrens farm than at the Quitman farm (Fig. 3.1). The Wrens farm lost as much as 60 kg N ha^{-1} in a given period, whereas the Quitman farm did not lose more than 2.7 kg N ha^{-1} in a sampling period. Overall ammonia losses during the study amounted to 32% of the applied N at the Wrens farm and 2% of the applied N at the Quitman farm. The higher ammonia loss from broadcast, granular urea at the Wrens farm is similar to losses found in other studies. In a two-year study with tall fescue pastures in north Georgia, Vaio et al. (2008) reported average losses of 25% of the applied N with broadcast, granular urea. Similarly, Sommer and Jensen (1993) in a study measuring NH_3 volatilization from urea and ammoniacal fertilizers applied to winter wheat and grassland reported NH_3 losses of up to 25% of applied N. The smaller NH_3 loss at the Quitman farm than at the Wrens farm may be attributed to the form of N applied (UAN vs. urea) and the application method used. The Quitman farm applied UAN through the irrigation system, whereas the Wrens farm broadcast granular urea. By applying the N through the irrigation system, it is likely that the UAN solution infiltrated deep enough below the soil surface to prevent significant ammonia losses. In contrast, when granular urea is broadcast, urea hydrolysis occurs at or near the soil surface, which enhances the potential for NH_3 loss.

Furthermore, conditions favorable to ammonia loss from urea, such as relative humidity greater than the critical relative humidity (CRH) for urea (Vaio et al., 2008), and low soil pH buffering capacities (Table 3.1), were present at the Wrens farm. The low pH buffering capacity of the soils, which is mainly due to the low organic C content (Weaver et al., 2004), makes the soils more susceptible to increases in pH caused by urea hydrolysis. As a result, the soil pH in the vicinity of urea granules can rise well above pH 7, which leads to the formation and subsequent gaseous loss of ammonia (Mundy, 1995).

Supplemental irrigation can play an important role in the rates of ammonia volatilization from N fertilizers (Denmead et al., 2004). Mundy (1995) found that adding water to the soil increased urea-N recovery in a pasture from 79 to 91% with 10 mm, and from 79 to 94% with 50 mm. This effect was probably due to leaching of urea into the soil, thereby reducing NH_3 losses. In a study with NH_3 volatilization from nitrogen fertilizers surface-applied to grass pasture, Oberle and Bundy (1987) indicated that a rainfall/irrigation event must occur within 24 h after applying urea to prevent significant loss. Thus, applying irrigation immediately after urea application to a pasture would be expected to incorporate urea into the soil, thereby reducing ammonia loss. The amount of rain or irrigation required to move urea into the soil in pastures is not well established (Denmead et al., 2004). Soil water content at the Wrens farm was maintained between 40 and 80% plant available water in the top 60 cm of soil by irrigation, but it is unknown how much time elapsed between fertilizer and irrigation applications. From the large losses of ammonia measured at the Wrens farm, it is likely that irrigation was not applied soon after urea applications.

Not all ammonia volatilization from pasture ecosystems is derived from the applied N fertilizers. Ammonia is also lost from N in manure and urine deposited on the surface of the pasture (Eckard et al., 2003), with urine being the larger of these two sources. Bussink (1992) found ammonia losses equivalent to approximately 8% of the total amount of excreted N. Furthermore, he found that the larger the amount of N fertilizer applied, the larger the N content found in cow excreta and the larger the loss of NH_3. Given the amount of N applied to the pastures at the Wrens farm, a significant fraction of the NH_3 loss may have been derived from cow excreta. Additionally, the application of poultry litter at the Wrens farm may have contributed to the measured ammonia losses (Marshall et al., 2001).

Nitrate Leaching

The maximum nitrate concentration in the cup lysimeters observed at the Wrens farm was 4.8 mg N L^{-1} (Fig. 3.2). This occurred in January, when water flux through a 1-m depth was small (3.14 cm). Large water fluxes were estimated in October/November and March/April, but nitrate concentrations in those months were low resulting in low amounts of leached NO_3 (0 to 0.5 16 kg N ha^{-1} mo^{-1}; Fig. 3.1). Total nitrate losses for the monitored period (June 2008 to August 2009) were 1.7 kg N ha^{-1} (Table 3.2), which correspond to 0.3% of the applied N. Total water flux at 1-m depth in the monitored period was 55.22 cm, which corresponds to 33% of the total amount of water (precipitation + irrigation) received.

Nitrate concentrations and water fluxes at a 1-m depth from the soil types at the Quitman farm (Tifton, Stilson, and Fuquay soils) are presented in Figures 3.3, 3.4, and 3.5. In general, nitrate concentrations and water fluxes were greater than those for the Wrens farm. Maximum concentrations ranged from 12 to 27 mg N L^{-1} with maximum water fluxes as high as 30 cm $month^{-1}$. As a result, leached NO_3 ranged from 1 to 16 kg N ha^{-1} mo^{-1} at the Quitman farm. Total nitrate leaching losses for the monitored period (August 2008 to August 2009) amounted to 40 kg N ha^{-1}, which corresponds to 12.5% of the applied N (Table 3.2). Total water flux at 1-m depth in the monitored period ranged from 88 to 95 cm in the different soils (Fig. 3.3, 3.4, and 3.5), which represented 44 to 49 % of the total amount of water (precipitation + irrigation) received. Nitrate concentrations measured at the Quitman farm were similar to those determined by Pakrou and Dillon (2004) in a study with grazed, unfertilized, irrigated, and non-irrigated paddocks on sandy loam soils. Mean concentrations in that study ranged from 13 to 17 mg N L^{-1} for irrigated paddocks, and from 9 to 11 mg N L^{-1} for non-irrigated paddocks. Their study suggests that soil moisture and the amount of water that passes through the soil profile are important factors in NO_3 losses.

In a study with grazed pastures on sandy soils, Di and Cameron (2002) estimated that 33 kg N ha^{-1} yr^{-1} were expected to be lost through leaching considering urine patches alone. When up to 400 kg N ha^{-1} y^{-1} (split into four applications) were applied on top of paddocks already affected by urine, a total of 30 to 60 kg N ha^{-1} yr^{-1} was leached. These losses (7.5 to 15% of applied N) are similar to losses observed on the Quitman farm and suggest that multiple fertilizations may reduce nitrate losses.

Nitrate losses measured at both farms are smaller than those found in other grassland systems. Jarvis (1993) measured NO_3 losses of 200 kg N ha^{-1} yr^{-1} from a grassland on well-drained soils, where 420 kg fertilizer N ha^{-1} was applied. Ledgard et al. (1999), working on N inputs and losses for a clover/grass pasture grazed by dairy cows, found NO_3 losses of 137 to 204 kg N ha^{-1} yr^{-1} with fertilizer inputs of 400 kg N ha^{-1} yr^{-1}. These results suggest losses of about 50% of the N applied as fertilizer. Korevaar (1992) also found similar results.

CONCLUSIONS

The largest loss of N at the Wrens Farm occurred through ammonia volatilization, whereas the largest loss of N at the Quitman Farm occurred through nitrate leaching. Large ammonia losses at the Wrens Farm were attributed to the use of granular urea applied on the soil surface without sufficient immediate irrigation to incorporate the fertilizer. Ammonia losses at the Quitman farm were relatively small because of the use of UAN through the irrigation system, which presumably allowed a deeper movement of the fertilizer into the soil. Thus, better management of the N fertilization at the Wrens farm, either by incorporating urea with irrigation or using UAN through the irrigation system, may reduce ammonia volatilization losses. With regard to nitrate leaching, the larger losses at the Quitman farm than at the Wrens Farm were attributed to larger water fluxes at the 1-m depth coupled with larger nitrate concentrations in the leachate. Thus, better management of irrigation at the Quitman farm may reduce nitrate leaching losses. Additional research should be conducted on pasture-based dairies to better understand their nitrogen cycling and if possible improve their efficiency.

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Table 3.1. Soil chemical and physical properties for the two pasture-based dairy farms monitored in this study.

Depth (cm)	Soil Characteristic	Wrens Farm	Quitman Farm		
		Orangeburg	Tifton	Fuquay	Stilson
0-25	mmol $H^+ kg^{-1}$ (pH unit)$^{-1}$	2.9	3.3	2.6	3.3
	pH	6.2	5.9	6.3	5.7
	Ks (cm/hr)	0.25	0.74	2.20	0.39
	Total C (g/kg)	6.8 - 8.9	6.7	6.1	7.6
	CEC ($cmol_c$/kg)	1.3	2.0	2.1	1.8
	Bulk Density (g/cm^3)	1.6	1.7	1.5	1.5
	Wilting Point (cm^3/cm^3)	0.027	0.014	0.017	0.021
	Field Capacity (cm^3/cm^3)	0.187	0.192	0.123	0.133
	Sand (%)	79.5-88.6	79.5	90.9	92.5
	Silt (%)	9.1-14.5	13.8	6.1	4.8
	Clay (%)	2.3-6.8	6.8	3.0	2.7
25-50	mmol $H^+ kg^{-1}$ (pH unit)$^{-1}$	3.8	4.9	3.4	2.4
	pH	5.7	5.7	6.1	5.4
	Ks (cm/hr)	0.22	0.17	1.03	0.51
	CEC ($cmol_c$/kg)	0.9	1.6	1.8	0.9
	Bulk Density (g/cm^3)	1.7	1.7	1.5	1.5
	Wilting Point (cm^3/cm^3)	0.049	0.034	0.034	0.049
	Field Capacity (cm^3/cm^3)	0.303	0.226	0.208	0.241
	Sand (%)	70.6-83.4	70.9	84.0	84.1
	Silt (%)	10.6-15.4	15.4	5.1	10.1
	Clay (%)	9.2-16.7	13.7	10.9	5.9
50-75	mmol $H^+ kg^{-1}$ (pH unit)$^{-1}$	3.9	4.6	4.3	3.8
	pH	5.5	5.0	4.9	4.6
	CEC ($cmol_c$/kg)	1.2	2.2	2.7	1.7
	Bulk Density (g/cm^3)	1.7	1.6	1.5	1.5
	Wilting Point (cm^3/cm^3)	0.050	0.064	0.070	0.073
	Field Capacity (cm^3/cm^3)	0.330	0.280	0.278	0.276
	Sand (%)	64.0-76.7	64.2	66.6	80.6
	Silt (%)	9.0-12.4	10.4	8.3	5.4
	Clay (%)	14.3-26.1	25.4	25.0	13.9
75 +	mmol $H^+ kg^{-1}$ (pH unit)$^{-1}$	4.1	4.9	4.0	4.4
	pH	5.6	5.2	5.1	4.8
	CEC ($cmol_c$/kg)	1.3	2.1	2.4	2.1
	Sand (%)	43.4-67.3	50.0	67.3	74.4
	Silt (%)	6.7-10.3	10.1	4.5	5.8
	Clay (%)	27.2-39.8	39.9	28.1	19.8

Table 3.2. Summary of total fertilizer N applied and losses associated with the sampling period at Wrens (1 Jun 2008-31 Aug 2009) and Quitman (1 Aug 2008-31 Aug 2009) Farms

	N Fertilizer Applied	Poultry Litter N	Ammonia Volatilized	Nitrate Leached
	---------- kg N ha^{-1} (% of Total N applied) -----------			
Wrens Farm †	605	134	196 (32)	1.7 (0.3)
Quitman Farm ‡	320	0	7 (2)	40.4 (12.5)

† Fertilizer N = Urea
‡ Fertilizer N = UAN

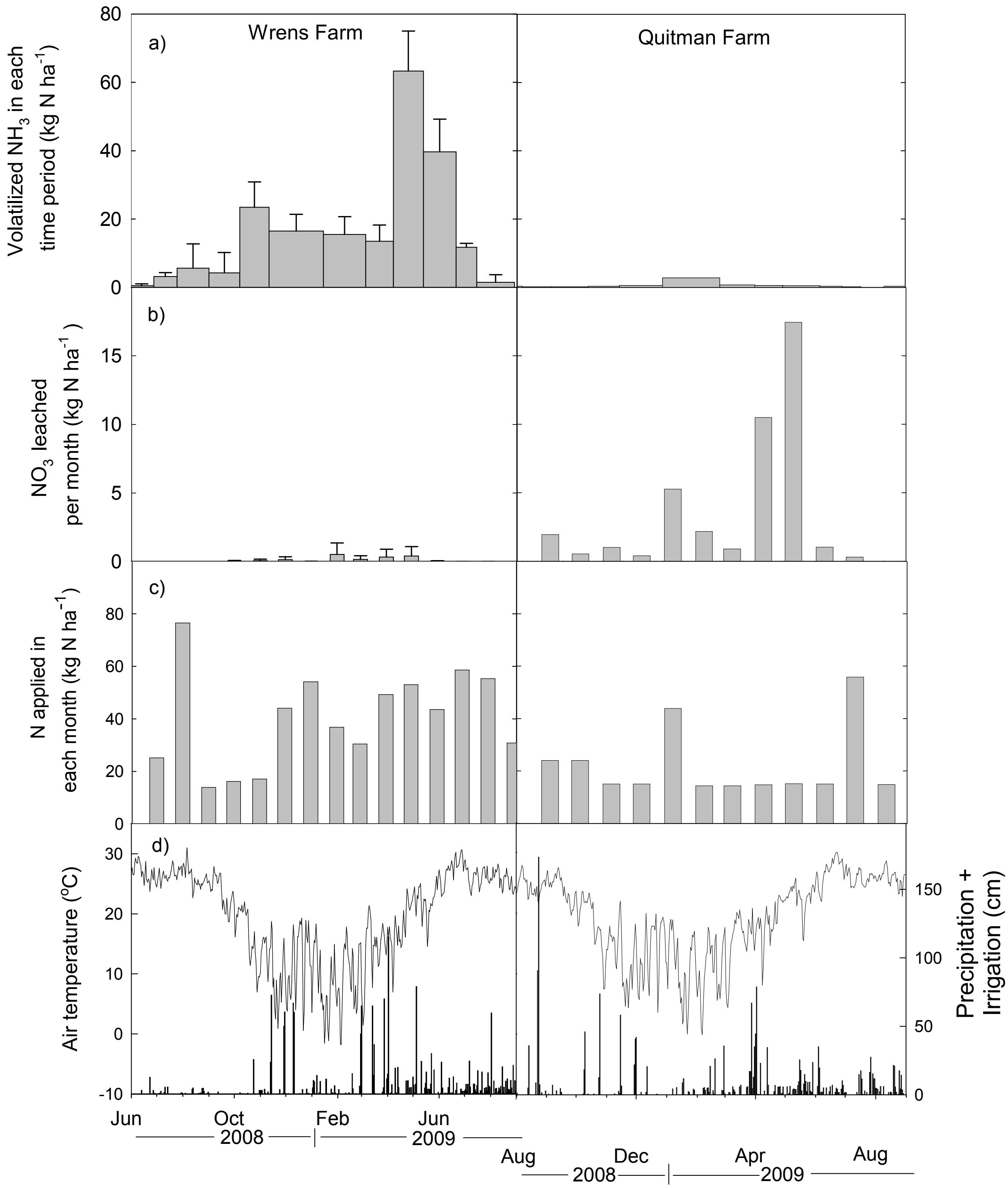

Fig. 3.1. Ammonia volatilization (a), nitrate leaching (b), fertilizer-N applied (c), and air temperature and precipitation (d) at each farm during sampling periods in 2008 and 2009.

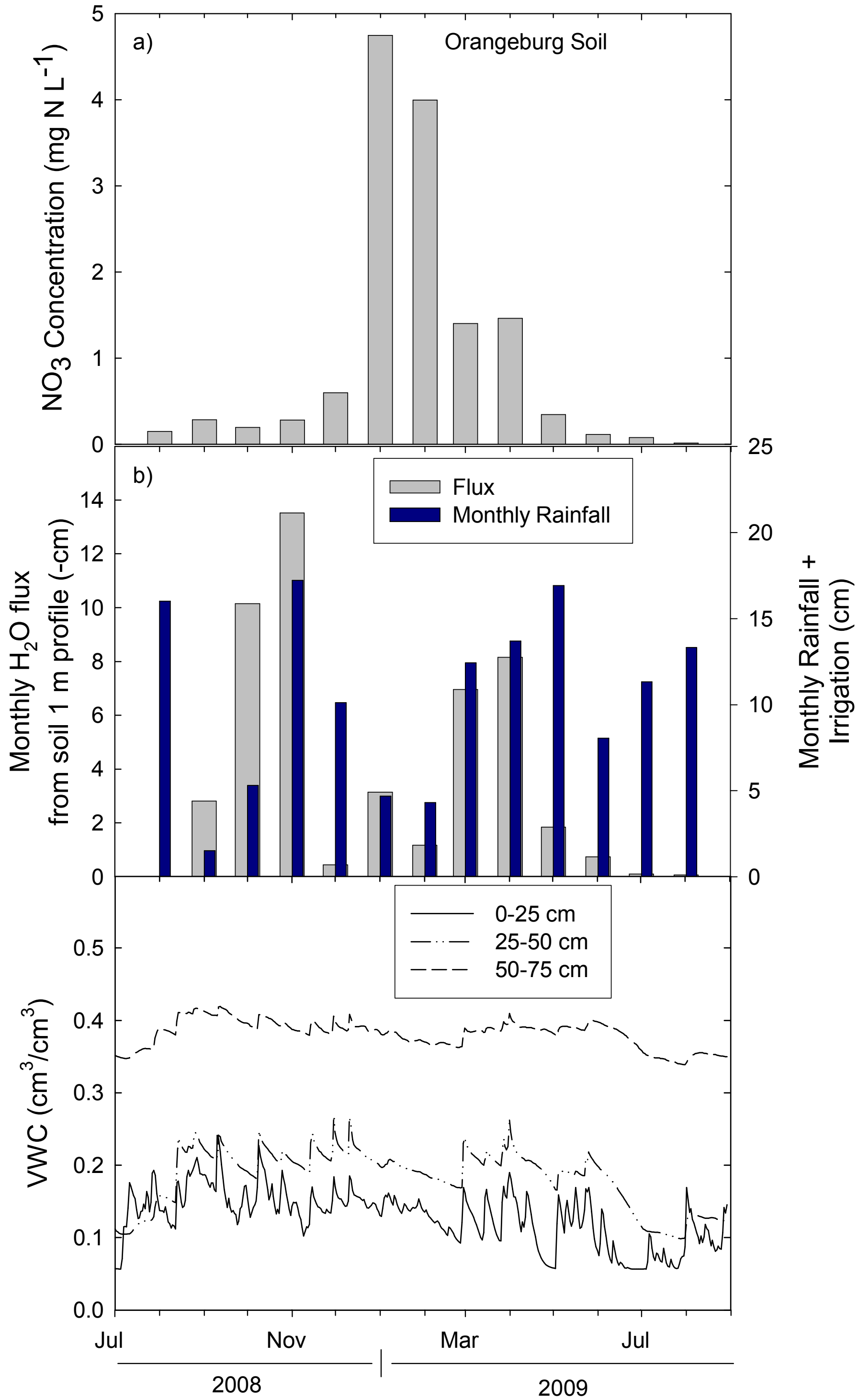

Fig. 3.2. Nitrate concentrations in 1-m cup lysimeters (a), water flux at 1-m depth (b), and volumetric water content at 0-25, 25-50, and 50-75 cm in the Orangeburg soil on the Wrens farm.

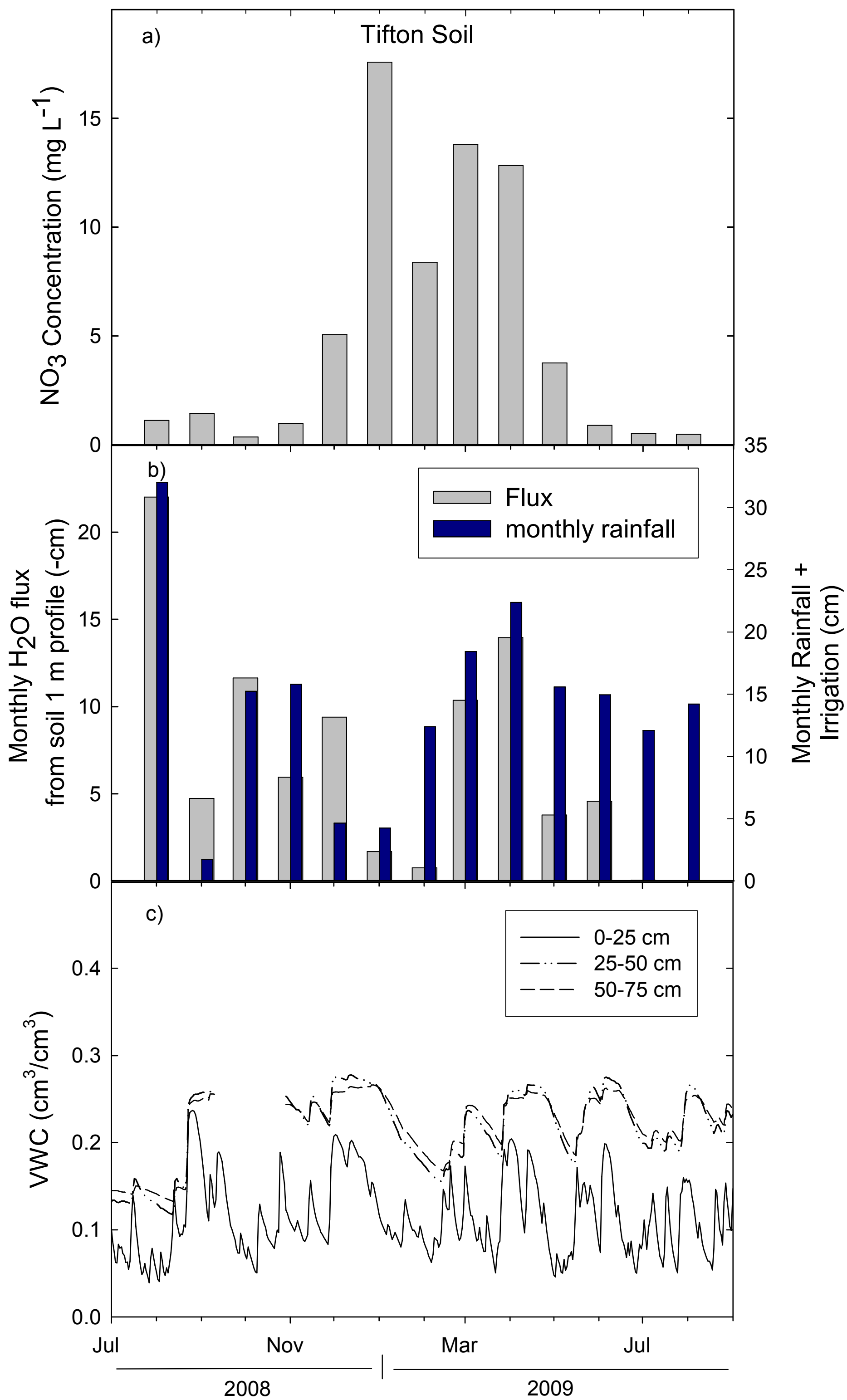

Fig. 3.3. Nitrate concentrations in 1-m cup lysimeters (a), water flux at 1-m depth (b), and volumetric water content at 0-25, 25-50, and 50-75 cm in the Tifton soil on the Quitman farm.

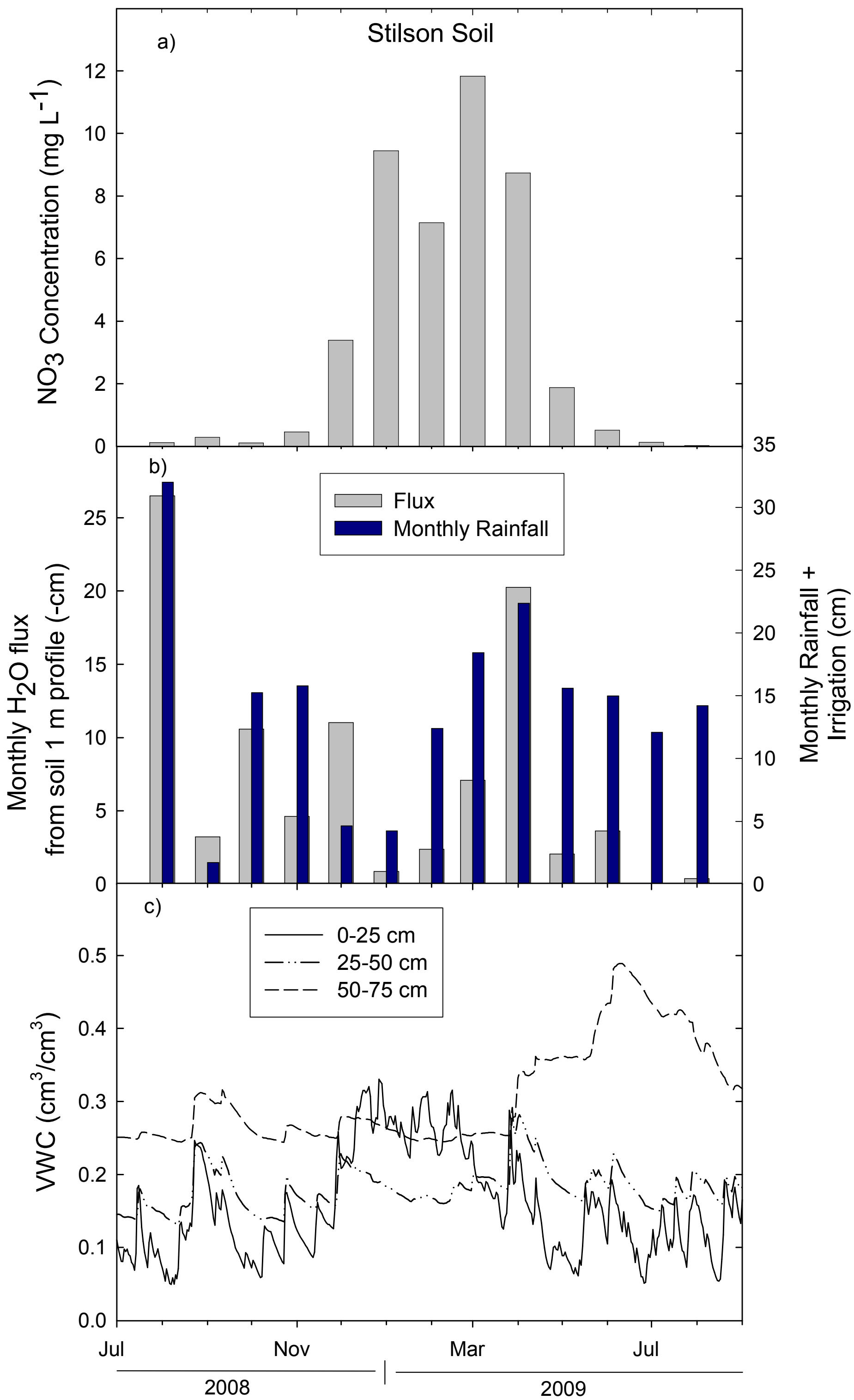

Fig. 3.4. Nitrate concentrations in 1-m cup lysimeters (a), water flux at 1 m (b), and volumetric water content at0-25, 25-50, and 50-75 cm in the Stilson soil on the Quitman farm.

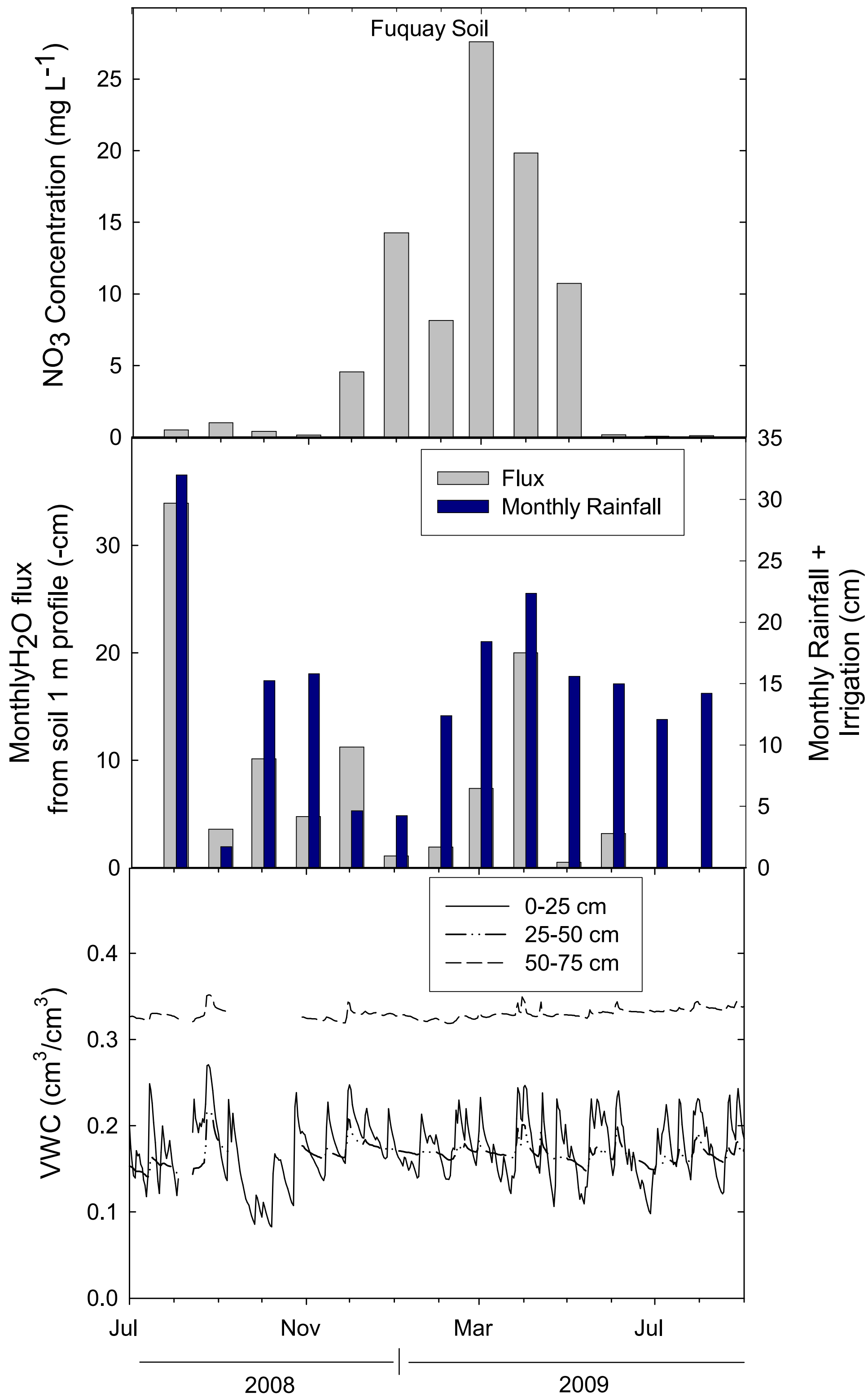

Fig. 3.5. Nitrate concentrations in 1-m cup lysimeters (a), water flux at 1 m (b), and volumetric water content at 0-25, 25-50, and 50-75 cm in the Fuquay soil on the Quitman farm.

CHAPTER 4

FARM-GATE N BALANCES FOR TWO PASTURE-BASED DAIRIES IN GEORGIA, USA

Eason, N., M.L. Cabrera, N. Hill, C. Mackowiak, and K. Roach. To be submitted to Nutrient Cycling in Agroecosystems

ABSTRACT

Management intensive pasture-based dairies are an alternative to traditional confinement dairy systems in the southeastern USA. The objective of this study was to develop a one-year, farm-gate N balance for two pasture-based dairies (Wrens and Quitman farms) located in the Coastal Plain of Georgia, USA. Nitrogen inputs were obtained from farm records and included fertilizer N applied, N in feed, hay, and silage imported to the farm. Nitrogen outputs included export of N in milk, nitrate leaching (NO_3), ammonia volatilization NH_3), and nitrous oxide (N_2O) emission. Cup lysimeters placed at 1-m depth were used to monitor NO_3 concentrations, and NO_3 leaching was calculated from modeled water fluxes. Ammonia volatilization losses were measured with a micrometeorological technique using passive ammonia samplers. Nitrous oxide emissions were measured using closed chambers in the field. Nitrate leached, expressed as % of total N inputs, amounted to 5.3 % at the Quitman farm and 0.3% at the Wrens farm. In contrast, NH_3 volatilization losses amounted to 0.9% at the Quitman farm and 30.2% at the Wrens farm. The larger NH_3 losses at the Wrens farm were attributed to surface applied urea compared to UAN applied through center pivot irrigation at the Quitman farm. Both farms had large surpluses of N, which was probably immobilized N. Total N inputs did not differ much between farms, but the form and management of N inputs affected losses and surpluses in each farm. Overall our results indicate that these pasture-based dairies have losses similar to other grassland systems.

INTRODUCTION

The number of pasture-based dairies is increasing in the Coastal Plain region of Georgia (USA) as an alternative to traditional confinement dairy systems. Pasture-based dairies rely on an intensive system of pasture management and utilization (Ledgard et al., 1999). Consequently, proper utilization of nutrients on these farms is vital to operational efficiency, profitability, and environmental impacts (Jarvis, 1993). Nitrogen fertilizer is the largest costs for these pasture-based dairies. Therefore, losses of N from these systems may have severe economic as well as environmental consequences. Jarvis (1993) stated that N in grasslands can be lost through NH_3 volatilization, NO_3 leaching, and N_2O emitted from different biophysical components of the system. A perspective of the relative importance of the forms of N losses in a pasture-based dairy can be gained using a farm-gate balance approach because it takes into account inputs and outputs from the farm. According to Nevans et al. (2006), farm-gate balances tend to be much simpler and have less uncertainties and unknowns in comparison to a particular soil-based balance. Furthermore, farm-gate balances are indicators of how a particular nutrient is moving through a system and can serve as a guideline for improving nutrient management.

Most studies on farm-gate balances have been conducted in Australia, New Zealand, and some European countries (e.g., Watson and Atkinson, 1999; Ball and Ryden, 1984; Ledgard et al., 1999; Ledgard, 2001; Humphreys et al., 2008; Nevans et al., 2005; Eckard et al., 2007, Korevaar, 1992). While those studies are useful to structure a farm-gate balance and to obtain estimates of expected N losses, data from those studies cannot be extrapolated to southeastern U.S. pasture-based dairies because of differences in soils and climatic conditions. Therefore, the objective of this study was to develop farm-gate N balances for two pasture-based dairies located in the Coastal Plain of Georgia, USA.

MATERIALS AND METHODS

Farm Characteristics

Two commercial pasture-based dairy farms were chosen as experimental sites for this study. One farm was located in east-central Georgia approximately 5 km west of the city of Wrens (Wrens farm); the second farm was located in South Georgia, approximately 25 km south of the city of Quitman (Quitman farm). The Wrens farm consisted of 124 ha and 500 cross-bred Jersey-Holstein milk cows, whereas the Quitman farm had 114 ha and 450 Jersey-Holstein milk cows. Both farms grew forages under center pivot irrigation systems, which were divided into paddocks approximately 2.5 ha in size. The dominant soil at the Wrens farm is an Orangeburg soil series (fine-loamy, Kaolinitic, thermic Typic Kandiudults) with class 0-3 and 3-7% slope classifications. The dominant soils at the Quitman farm include Tifton (fine-loam, siliceous, thermic Plinthic, Argillic), Fuquay (loamy kaolinitic, thermic arenic plinthic Kandiudults), Stilson (loamy sand, Siliceous, subactive, thermic arenic plinthic Paleudults), and Alapaha series (loamy, siliceous, subactive, thermic arenic plinthic Paleaquults). Soils at both farms are loamy sand in the A_p horizon (25 cm) and change to sandy clay loam in the B_t horizon.

Perennial pastures at the Wrens farm were planted to ‘Tifton 85’ bermudagrass (16 ha) (*Cynodon dactylon (*L.*) Pers.*) or ‘Jesup Max Q’ tall fescue (*Lolium arundinacea Schreb*), (23 ha). Approximately 59 ha of prepared ground was planted to ‘Tifleaf 3’ pearl millet (*Pennisetum glaucum (*L.*) R. Br.*) in the spring and over seeded with a mixture of ‘Wrens Abruzzi’ cereal rye (S*ecale cereal (*L.*) Salisb.*) and ‘Feast 2’ annual ryegrass (*Lolium multiflorum Lamarck*) in September. Bermudagrass pastures were also over-seeded with cereal ryegrass. Tall fescue pastures were over-seeded with annual ryegrass in the fall of the year. The forage system at the

Quitman dairy was 'Tifton 85' bermudagrass which was over seeded with 'Florida 401' cereal rye and 'Big Boss' annual ryegrass in November. At the Wrens farm, N fertilizer was broadcast in the form of granular urea with a minor amount supplied as chicken litter (134 kg ha^{-1}). At the Quitman farm, N was applied as urea ammonium nitrate (UAN) through the center pivot irrigation system.

Nitrogen Balance

The farm-gate N balance considered three main N sources as inputs on the farm: a) nitrogen fertilizer, b) N imported forages (corn silage and hay), and c) imported grain feed. Outputs of N from the farms included: a) NH_3 volatilization, b) N_2O emissions, c) NO_3 leaching, and d) milk N export. Inputs that were not considered in the balance were atmospheric N deposition and N fixation by legumes. Atmospheric deposition was not considered because it usually contributes very little to overall inputs (Ball and Ryden, 1984). Nitrogen fixation by legumes was not taken into account because legumes were not a significant component of the forage species on either farm. Output of N in the meat of culled animals was not taken into account because very few cows were culled during the study. The farm-gate N balance (surplus) was calculated as inputs minus outputs (Fig. 4.1).

Soil Sampling

Soils were sampled with a hydraulic probe in 25-cm increments to a depth of 100 cm in June 2008 and August 2009 at both farms. Six paddocks at the Wrens farm were sampled at eight randomly selected locations within each paddock. These paddocks represented the main soil type or pasture species present on the farm. Four paddocks were sampled in a similar manner at the

Quitman farm, one paddock for each soil type. Samples from the respective depths in each paddock were combined, air dried, and ground to pass a 2-mm screen using a Wiley mill. Soils were extracted for NO_3 and NH_4 and analyzed colorimetrically on an Alpkem auto-analyzer (Mulvaney, 1996).

Managed Inputs and Outputs

Farm Records

Farm records were used to determine N inputs and outputs that were a direct result of farm management. Records from the Wrens farm provided the following information: a) number of milk cows grazing in a given week and weekly grazing rotational period, b) weekly estimates of forage dry matter (DM) in each paddock, c) average amount of feed (hay or grain) given to milk cows in the milking parlor on a weekly basis, d) amount of fertilizer N applied on the farm, and e) daily export of milk. Records from the Quitman provided monthly estimates of a) grain and hay feed, b) milk export, and c) amounts of UAN fertilizer applied. To estimate N contents (kg N) of feed, forage, and milk, crude protein values were divided by constants (6.25 for forage, 6.25 for hay, 5.7 for grain, 6.38 for milk) to obtain percent N, which was then multiplied by quantities of each source. Crude protein was obtained from farm records for milk, from laboratory analysis for forages, and from book values for hay and corn silage.

Outputs in Environmental Losses

NH_3 Volatilization

Ammonia volatilization was measured using a micrometeorological passive flux sampler method (Sommer et al., 1994). Samples were collected every 30 to 60 days as described in Chapter 3. Ammonia volatilization was measured from the areas under two center pivot irrigation systems on the Wrens farm and from under one center pivot on the Quitman farm. Ammonia losses were expressed as a monthly loss (kg N ha^{-1}) by proportionally allocating the amount of ammonia volatilized in a given sampling period to the days corresponding to each month included in that particular period. At the Wrens farm, the rate of loss (kg N ha^{-1} mo^{-1}) for each pivot was then multiplied by the area under each pivot and the results added to obtain total monthly losses from the pivots. At this farm, 21 ha were not a part of the two pivots which were monitored for NH_3 losses, but because that land was treated similarly to the pivots, an average volatilization rate was calculated between the two pivots and was multiplied by 21 ha to obtain additional losses from that area. At the Quitman farm, volatilization was measured from only one pivot, so the rate of loss from that pivot was multiplied by the total area of the farm to obtain total losses. Grazed areas that were not under the monitored pivot were treated similarly as the grassland under the pivot.

Nitrous Oxide Emissions

Nitrous oxide is emitted from cattle urine and manure by microbial metabolism of the N sources found in each (Luo et al., 2008). Depending upon the N source, the rate and extent of N_2O emissions vary. To account for the differences based upon the N source (urine vs. manure), a field study was carried out on the Wrens farm. Before conducting the field study, a preliminary laboratory experiment was carried out to determine an approximate sampling regime to be used in the field. Air dried soil was packed to field bulk density into 18 closed-ended PVC chambers (16-cm I.D. x 15.5 cm height). Two replicates of each treatment (manure, urine, and control) were placed in incubators at 15, 25, or 35^{o}C for two weeks. Manure and urine treatments received surface applications of 159 g of fresh bovine manure and 88 mL of bovine urine, respectively. Every day for 1 h, the chambers were capped with PVC caps equipped with septa for gas sampling. A 3-mL sample was withdrawn with a hypodermic needle and syringe at time zero and after 1 h. The cap was removed and the same procedure was repeated daily for two weeks. From that study, it was determined that a suitable sampling regime for the field study was to take gas samples on the 0, 2, 4, 7, 10, and 14 days after manure or urine application. Two 14-d sampling periods during the summer of 2009 (starting on 15th, June 2009 and on 13th, July 2009) were used for the field study. Chambers were installed (in a paddock that had been grazed the previous day) in a grid pattern with six rows (blocks) spaced approximately 6 m from each other, and three chambers spaced approximately 6 m within each row. The chambers were inserted 7.75 cm into the ground while leaving 7.5 cm for headspace (in addition, 3 cm from the cap). Three treatments: urine, manure, and a control were randomly assigned in each of six blocks. Manure treatments received 400 g fresh manure (870 g H_2O kg^{-1}, 53.76 g N m^{2}) and urine treatments received 88 mL of urine (9.4 g N m^{2}). The same sampling method used in the lab was used in the field, but with the new sampling regime. Samples were analyzed with a Varian gas chromatograph (STAR 3600 CX, Palo Alto, CA) for N_2O and CO_2 concentrations. Nitrous oxide and CO_2 concentrations at each sampling were integrated with time using the trapezoidal rule to estimate total emissions in 14 days (Table 4.3). The first study was carried at a relatively low WFPS (18 to 20%), whereas the second study was conducted at a relatively high water-filled pore space (WFPS) (30 to 60%). Therefore, the total N_2O emission in 14 d from each study was used to develop a moisture correction factor (MF) based on WFPS:

MF (0-1) = 1-0.0183*(Average WFPS during 14-day) [1]

A correction factor (TF) based on average temperature during 14 d (^oC) was calculated as follows (Li et al., 1992):

TF (0-1) = $2^{((\text{Avg. 14 day soil temp}-28.7)/10)}$ [2]

Dataloggers (Model CR206X, Campbell Scientific, Inc., Logan, UT) connected to time domain reflectometry sensors (CS625) and temperature probes (Model 109) were used to record soil water content and temperature every 15 min in the upper 25 cm of soil at three locations in the Wrens farm and at four locations at the Quitman farm. Fourteen-day (estimated rotational time) averages of soil water content and temperature were combined with the total emission measured (kg N ha^{-1} in 14 d) from each of the treatments (manure, urine, control) in the first study, to calculate a corrected emission every 14 d throughout the year for each of the treatments:

Corrected N_2O emission in 14-d (kg N ha^{-1}) = Measured kg N ha^{-1}14 d x TF x MF [3]

To scale the N_2O emissions to a whole-farm scale, we calculated the percent area of each hectare that was affected by urine and manure by taking into account that 250 cows grazed a paddock in half a day, and by assuming an average manure patch affects 0.12 m^2 and an average urine patch affects 0.36 m^2 (White et al., 2001). The total emission of N_2O from a given hectare was estimated by using the corrected emission rates during 14 d for areas affected by manure and urine, as well as for areas not affected by the cattle excretions, and multiplying them by the corresponding percentages of a hectare affected by the different treatments. Assuming an average rotation period of 14 d, each hectare on the farm was grazed by cattle every 14 d.. These calculations were carried out for the Wrens farm, which was where the field study on N_2O emission was carried out. Nitrous oxide emissions for the Quitman farm were not calculated.

Nitrate Leaching

Nitrate leaching was measured using five cup lysimeters (representing one station) placed on each soil type on each farm. Each cup lysimeter was 1 m in length and was installed 3 m from one another along an electric fence-line to protect them from the cows. The lysimeters were placed under a -52 kPa of vacuum with water samples collected bi-weekly and analyzed colorimetrically for NO_3 on an Alpkem auto-analyzer (Mulvaney, 1996). The results were expressed in kg N $ha^{-1}mo^{-1}$. Because the Wrens farm had only the Orangeburg soil type, NO_3 losses were estimated by calculating the average loss from three lysimeter stations and multiplying it by the total number of hectares on the farm. Because the Quitman farm had multiple soil types, losses calculated with each soil type were multiplied by the estimated area occupied by that soil on the farm (NRCS, 2009). The NO_3 losses were then added together for a total loss from the farm.

RESULTS AND DISCUSSION

Managed Inputs and Outputs

Managed inputs included fertilizer, grain, corn silage, and hay used. Rates and total amounts of N fertilizer varied between farms. At the Wrens Farm, monthly N application rates ranged from 16 to 55 kg N ha $^{-1}$ (Fig. 4.4) for a total yearly rate of 491 kg N ha $^{-1}$ (Table 4.1). At the Quitman Farm, monthly N applications ranged from 14 to 43 kg N ha $^{-1}$ (Fig. 4.4), for a comparatively smaller yearly rate of 297 kg N ha $^{-1}$ (Table 4.1). Consequently, fertilizer N as % of total N inputs represented 79% at the Wrens farm and 41% at the Quitman farm. The smaller rate of N fertilizer applied at the Quitman farm was in part counterbalanced by a larger rate of N input in the form of grain feed (383 kg N ha^{-1} versus 97 kg N ha^{-1} at the Wrens farm). Grain feed as % of total N inputs represented 16% at the Wrens farm and 53% at the Quitman farm. Thus, grain feed and fertilizer N accounted for most of the N input on both farms (95% at the Wrens farm and 94% at the Quitman farm). Hay and silage made up the rest of the N inputs (Table 4.1).

Monthly N exported in milk ranged from 574 to1,526 kg N ha $^{-1}$ for the Wrens farm, and from 405 to 956 kg N ha $^{-1}$ for the Quitman farm (data not shown). Total exported N in milk as a percentage of total N inputs was 15.8% for the Wrens farm and 17.1% for the Quitman farm (Table 4.1). Thus, both farms used similar proportions of the input N to produce milk. Ledgard et al. (1999) found that milk export averaged 20% of the total N inputs. Eckard et al. (2007) along with studies reported by Rotz et al. (2005) found higher milk exports ranging from 27 to 30% of the N inputs. Different concentrates, supplements, forage type and quality, and dietary management are likely key contributors to these differences.

Outputs in Environmental Losses

N_2O Emissions

In the field study carried out at the Wrens farm, N_2O emissions followed distinct patterns for each of the treatments. Emissions were greater from chambers containing urine and manure than from control chambers (Fig. 4.2 and 4.3), with the larger rates of emissions occurring within the first 2 days after application. Although urine and manure emission rates peaked between 4 and 5 mg N_2O m^2 hr^{-1}, overall emission rates remained at a lower rate for the majority of the study period. Luo et al. (2008) also saw emissions reach a maximum 1 to 14 days after grazing, thereafter returning to pre-grazing emission rates. Carbon dioxide emissions were greater for the manure treatment than from the urine or control treatments for two days following application, but were not different from the control treatments thereafter. Between days 7 and 10, CO_2 emission was smaller from the urine treatment than from the manure or control treatment. Estimating the total loss of N_2O-N over two weeks for a grazed paddock resulted in 1.78 and 0.78 kg N ha^{-1} for Studies 1 and 2, respectively. In both studies, total N_2O emission was greater from urine and manure treatments than from control treatments (Table 4.2). Furthermore, in the second study, emission from urine was greater than emission from manure. This may have been due to more anaerobic conditions created by the manure application, as indicated by a greater emission of CO_2. Carter (2007) suggests that nitrification and denitrification seem to contribute equally to the N_2O production in urine-affected soil. This could relate to the intermediate soil water content of about 45% WFPS, which would allow both processes to occur simultaneously.

Many studies have found good correlation between N_2O emission rates and water WFPS (e.g. Carter, 2007; Luo et al., 2008; Koops et al., 1997; Vinther, 2005). The second study in this experiment had lower overall emissions, but a higher WFPS, which may have been due to the fact that under wetter conditions a larger proportion of the emitted N is converted to N_2 gas rather than to N_2O (Vinther, 2005; Carter, 2007). In addition, an increase in WFPS would decrease nitrification, decreasing N_2O emissions. Because the studies were located in different paddocks, it is possible that different soil characteristics, such as pH, bulk density, etc. could have played a role in the different results of both studies. Turner et al. (2008) studied the variability of N_2O emissions on an irrigated dairy pasture, and concluded that, in addition to WFPS, topography, and NO_3 concentrations seemed to be the most influential factors. Topography indirectly affects N_2O emissions because the soil physical properties, structure, and water status effect the soil environment for denitrifying microorganisms.

Of the total inputs of N on the Wrens farm, N_2O represented a 2.65% loss from the system (3.36% of the total applied fertilizer), with estimated monthly losses ranging from 0.52 to 3.13 kg N ha^{-1} (Table 4.1). This represented a loss of 16.5 kg N ha^{-1} for the year. Ledgard et al. (1999) found annual losses of 15 to 30 kg N ha^{-1} on a farm receiving 400 kg N ha^{-1} with a stocking density of 4.4 cows ha^{-1}. Eckard et al. (2003) found N_2O losses ranging between 13 and 15 kg N ha^{-1} yr^{-1}on a temperate perennial grass and clover dairy pasture receiving 200 kg N ha^{-1} yr^{-1}. Thus, our results seem to parallel other N_2O studies with similar conditions. Compared to other outputs in the N balance, N_2O was second to least of the outputs from the system. Eckard et al. (2003) suggest better use of N management practices could significantly reduce N_2O emissions; however, from the point of view of N use efficiency it seems that one would focus on management practices relevant to outputs with the highest losses of N.

NH_3 Volatilization

Monthly ammonia losses ranged from 0.5 to 41.6 kg N ha $^{-1}$ at the Wrens farm, and from 0.1 to 1.5 kg N ha $^{-1}$ at the Quitman farm (Fig. 4.5). When expressed as percentage of the total N inputs, total ammonia loss was 30.2% at the Wrens farm and 0.9% at the Quitman farm (Table 4.1). Greater losses at the Wrens farm were likely due to the use of surface applied granular urea, in contrast to the Quitman farm, which used UAN solution through the irrigation system. The poultry litter that was applied to the Wrens farm may have also been a cause of the higher losses of NH_3 (Marshall et al., 2001). In research with surface-applied urea to fescue pastures in Georgia, Vaio et al. (2008) found losses of ammonia as high as 46% of applied N. In a summary of NH_3 volatilization results from many grazing dairy studies located in the UK, Europe, New Zealand, Netherlands, and Australia, Denmead et al. (2004) found a range of 25 to 60 kg NH_3-N ha $^{-1}$. Eckard et al. (2007) reported NH_3 emissions ranging from 45 to 74 kg N ha $^{-1}$ yr $^{-1}$ on pasture-based dairies receiving 200 kg N ha $^{-1}$ yr $^{-1}$. Similar losses were also found in other studies (Eckard et al., 2003; Ledgard, 2001). Average temperature and relative humidity for the state of Georgia (USA) and for countries with comparable ammonia studies can be found in Table 4.2. In general, Georgia's higher temperatures and relative humidity would be expected to results in larger NH_3 volatilization fluxes under intensive grazing conditions.

NO_3 Leaching Losses

Monthly nitrate losses ranged from 0 to 0.5 kg N ha $^{-1}$ for the Wrens farm and from 0 to 15.5 kg N ha $^{-1}$ for the Quitman farm (Fig. 4.6). Annual losses were estimated at 1.6 kg N ha $^{-1}$ for the Wrens farm and at 38.3 kg N ha $^{-1}$ for the Quitman farm (Table 4.1), which correspond to 0.3% and 5.3% of total N inputs, respectively. Larger water fluxes from the soil profiles, paired with significant NO_3 concentrations were the driving factors for NO_3 losses on both farms (Chapter 3). Nitrate concentrations were highest on both farms between December and April. Although water fluxes during those months were not consistently high, losses from these months were still the main contributors to NO_3 loss, with March and April being the highest. These elevated concentrations of NO_3 during the winter were expected and is explained by slower plant growth during the winter, and lower relative efficiency of N uptake by plants. Typically cool-season forages have shallower root depths compared to warm season forages (Ball et al., 2002). This coupled with higher rainfall during these months may leach NO_3 past the rooting zone. The Quitman farm also received 40 cm (irrigation and precipitation) more than the Wrens Farm during the year. It is interesting to note that although N fertilization at the Wrens farm was not reduced during winter, NO_3 leaching losses were low during winter. Studies on pasture-based dairies with similar fertilizer applications and stocking rates as the ones on this study found nitrate leaching losses ranging from 3 to 50% of the total N applied (e.g. Eckard et al., 2004; Eckard et al., 2007; Jarvis, 1993; Ledgard et al., 1999). Variability among studies can be attributed to differences in the soil type, climate, and timing of N application. Yearly and seasonal variations in weather patterns make management decisions for N applications difficult for farmers. Eckard et al. (2004) suggest that during low drainage years, NO_3-N may accumulate in the soil profile, contributing to higher losses in subsequent higher drainage years. To account for build-up of N in the soil, Ball et al. (2002) suggest frequent soil testing to optimize forage growth and N efficiency.

Nitrogen Surplus

The N balances from both farms resulted in a surplus of unaccounted for N. Surplus N found on the Wrens farm was 318 kg N $ha^{-1}yr^{-1}$ or 52% of the total input N. On the Quitman farm, surplus N was 559 kg N ha^{-1} or 77% of total N inputs (Table 4.1). Similar results were found by Nevans et al. (2006) in a compilation of data from intensively managed dairy farms across Europe. Their results showed N balance surpluses (input N – output N) of 378 and 238 kg N ha^{-1} in two separate studies.

Nitrogen surplus may include N in animals exported from the farm (culled cows and bull calves), increases in soil inorganic N, N losses in surface runoff, and N immobilized in organic fractions. Nitrogen amounts in animals exported from the farm and N losses in runoff were not estimated, but were probably small. Increases in inorganic N could not account for N surpluses because in both farms the amount of inorganic N in the soil decreased during the study. At the Wrens farm, the area-weighted inorganic N changed from 98 kg N ha^{-1} in the initial sampling to 55 kg N ha^{-1} in the final sampling, for an average loss of 43 kg N ha^{-1}. Similarly, at the Quitman farm, the area-weighted inorganic N changed from 65 kg N ha^{-1} in the initial sampling to 56 kg N ha^{-1} in the final sampling, for an average loss of 9 kg N ha^{-1} (Table 4.4). Therefore, immobilized N was the main suspected source of unaccounted N.

Studies in Denmark, discussed in Korevaar (1992), found that 85% of the N that entered the farm was lost to the system or accumulated in the soil. Ledgard et al. (1999) suggested that 16 to 30% of the added N could be immobilized into the soil as organic N after one year of N applications. Furthermore, the authors suggested that eventually much of the N immobilized would be mineralized in later years and be available to plants until these processes in the N cycle reached some type of equilibrium. Nevans et al. (2006) omitted net mineralization from their N balance because they assumed that mineralized soil N is replaced by immobilized N from newly added organic material (mainly through the input of manure). While most research agrees that this soil N steady-state assumption exists, Ball and Ryden (1984) describe it as being asymptotic; the time it takes the system to reach this equilibrium is unknown (Ball and Ryden, 1984; Jarvis, 1993; Ledgard et al., 1999; Watson and Atkinson, 1999). Thus, it would be likely to see higher rates of immobilization in the first few years of a pasture-based dairy than in later years. The two farms in the current study were less than two years old; therefore a relatively high percentage of N inputs being immobilized is possible. Because of the complexity of nitrogen cycling, understanding key controlling processes of mineralization/immobilization in grassland systems continues to be a difficult challenge (Ledgard, 2001; and Rotz et al., 2005). Further research into N cycling on these farms is warranted in order to understand the role of developing/developed pools of immobilized N on mineralized N over time.

CONCLUSIONS

Although both farms were operated as management intensive pasture-based dairies, there were differences in farm management. Total N inputs into each farm were similar, but the sources were different. Nitrogen fertilizer represented 79% of total N inputs at the Wrens farm, but only 41% of N inputs at the Quitman farm. In contrast, grain feed N as percentage of total N inputs was 16% for the Wrens farm and 53% for the Quitman farm. Nitrogen in milk exports was similar in both farms (16 to 17% of N inputs), but NH_3 loss was much larger at the Wrens farm (30% of N inputs) than at the Quitman farm (<1%). These differences in volatilized NH_3 were due to the type and method of fertilizer N used. Whereas the Wrens farm used surface-applied, granular urea, the Quitman farm used UAN solution applied through the irrigation system. As a result of the large differences in NH_3 losses, surplus N was greater at the Quitman (77% of N inputs) than at the Wrens farm (52% of N inputs). Most of the surplus N was probably immobilized in the soil, or present in the form of undecomposed fecal matter. Our results show that environmental and soil conditions as well as management practices play important roles in the transfer of N on these intensively grazed systems. Additional research should be carried out on the individual processes of N transfer in these systems to better identify the forms in which surplus N is present.

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Table 4.1. Farm-gate N balances for Wrens and Quitman farms.

	Wrens Farm	Quitman Farm	Wrens farm	Quitman Farm
	------------kg N yr^{-1} -----------		% of Total N input	
N input				
Mineral Fertilizer	60,395	33,807	78.9	40.7
Grain Feed	11,959	43,680	15.6	52.6
Maize Silage	1,224	0	1.6	0
Hay	3,011	5,587	3.9	6.7
Total Inputs	76,589	83,074	100	100
N output				
Milk	12,066	14,218	15.8	17.1
N_2O emission	2,032	N/A	2.7	N/A
NH_3 volatilization	23,104	746	30.2	0.9
NO_3 Leached	205	4,364	0.3	5.3
Total Outputs	37,407	19,328	48	23
N surplus	39,185	63,746	52	77

Table 4.2. Ranges in average yearly temperature and relative humidity for Georgia (USA) and other locations with studies on pasture-based dairies.

Location	Temperature (^{o}C)	Relative Humidity (%)
UK	2-13	70-85
Netherlands	(-1)-22	67-88
New Zealand	5-20	67-78
Australia	0-28	35-64
Denmark	(-3)-22	68-88
USA, Georgia	7.2-32	48-93

http://www.watkinshire.co.uk/assets/File/uk-weather-average-data.pdf (UK weather cite)

http://www.climatetemp.info/new-zealand/ NZ

Table 4.3. Mean N_2O emissions from manure, urine and, control treatments from a 14-day study at the Wrens farm in 2009.

Treatment	N_2O-N kg N ha^{-1} (14 d)$^{-1}$	CO_2-C kg C ha^{-1} (14 d)$^{-1}$
	----------------Study 1-----------------	
Manure	4.4 a	1231 a
Urine	4.3 a	913 b
Control	1.7 b	1372 a
	----------------Study 2------------------	
Manure	3.7 b	1991 a
Urine	5.7 a	1250 b
Control	0.7 c	1465 b

* Within a column, means followed by the same letter are not significantly different according to Fisher's protected LSD at a 0.05 probability level.

Table 4.4. Inorganic N to a depth of 75 cm at the initial and final samplings on the Wrens and Quitman farms.

Farm and Sampling Location	% of Farm area	Initial Sampling (June 2008)	Final Sampling (August 2009)	Change in soil N
		------------------------kg N ha^{-1}	-------------------------	
Wrens 1	16.7	73	49	24
Wrens 2	16.7	100	65	34
Wrens 3	16.7	127	44	83
Wrens 4	16.7	118	38	80
Wrens 5	16.7	63	97	-34
Wrens 6	16.7	108	36	73
Area Weighted Average		98	55	43
Quitman 1	37.5	72	64	8
Quitman 2	37.5	55	54	1
Quitman 3	5	54	41	14
Quitman 4	12	78	51	28
Quitman 5	5	103	63	40
Area Weighted Average		65	56	9

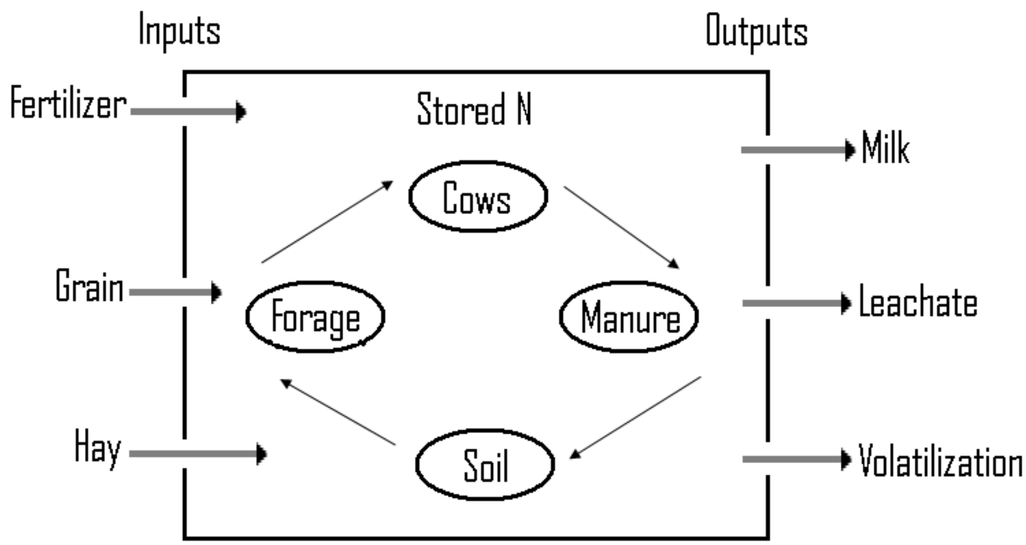

Fig. 4.1. Farm-gate balance: nitrogen inputs and outputs used.

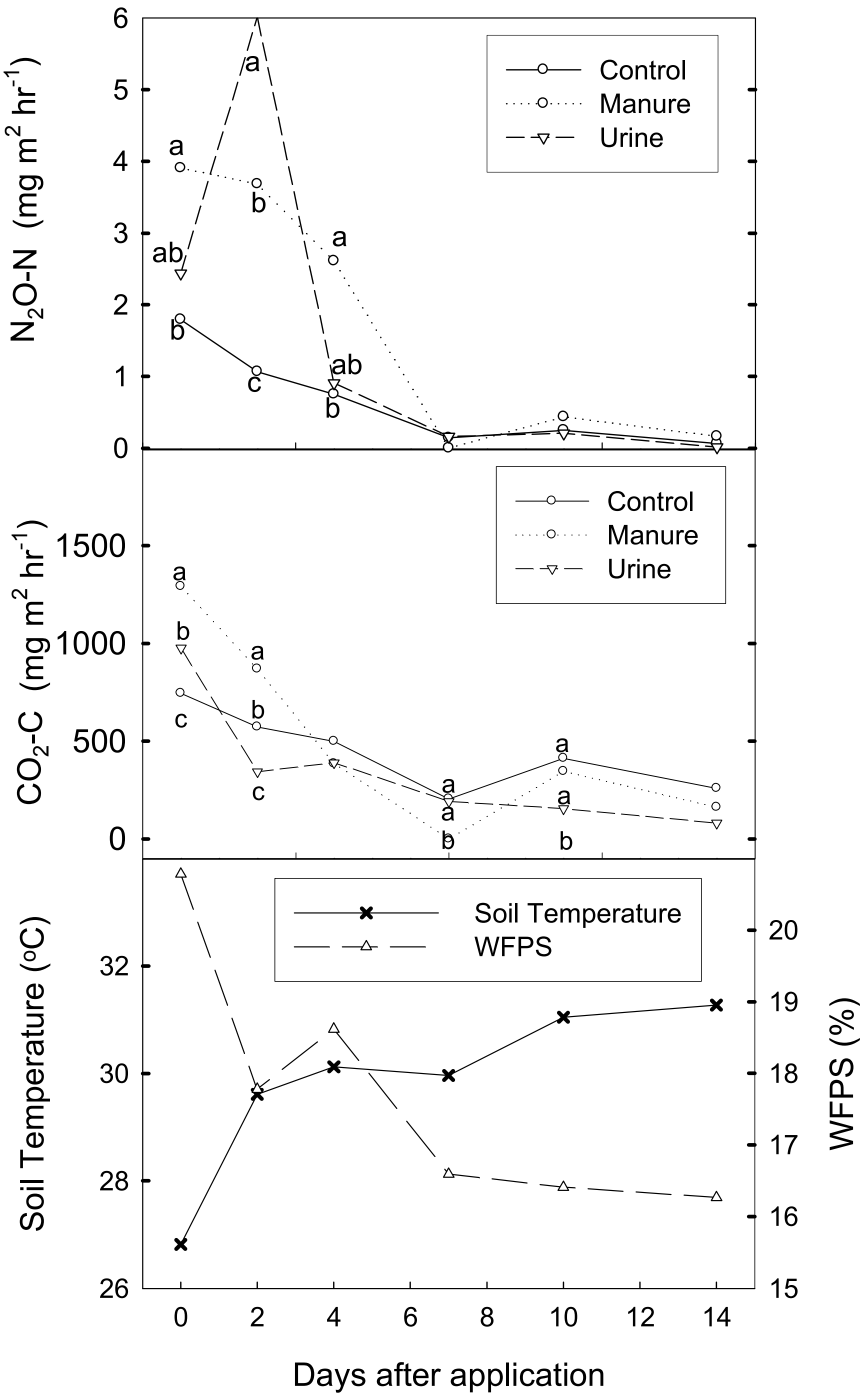

Fig. 4.2. Nitrous oxide and CO2 emissions, soil temperature, and water-filled pore space of the upper 10 cm during the first N_2O study on the Wrens farm.

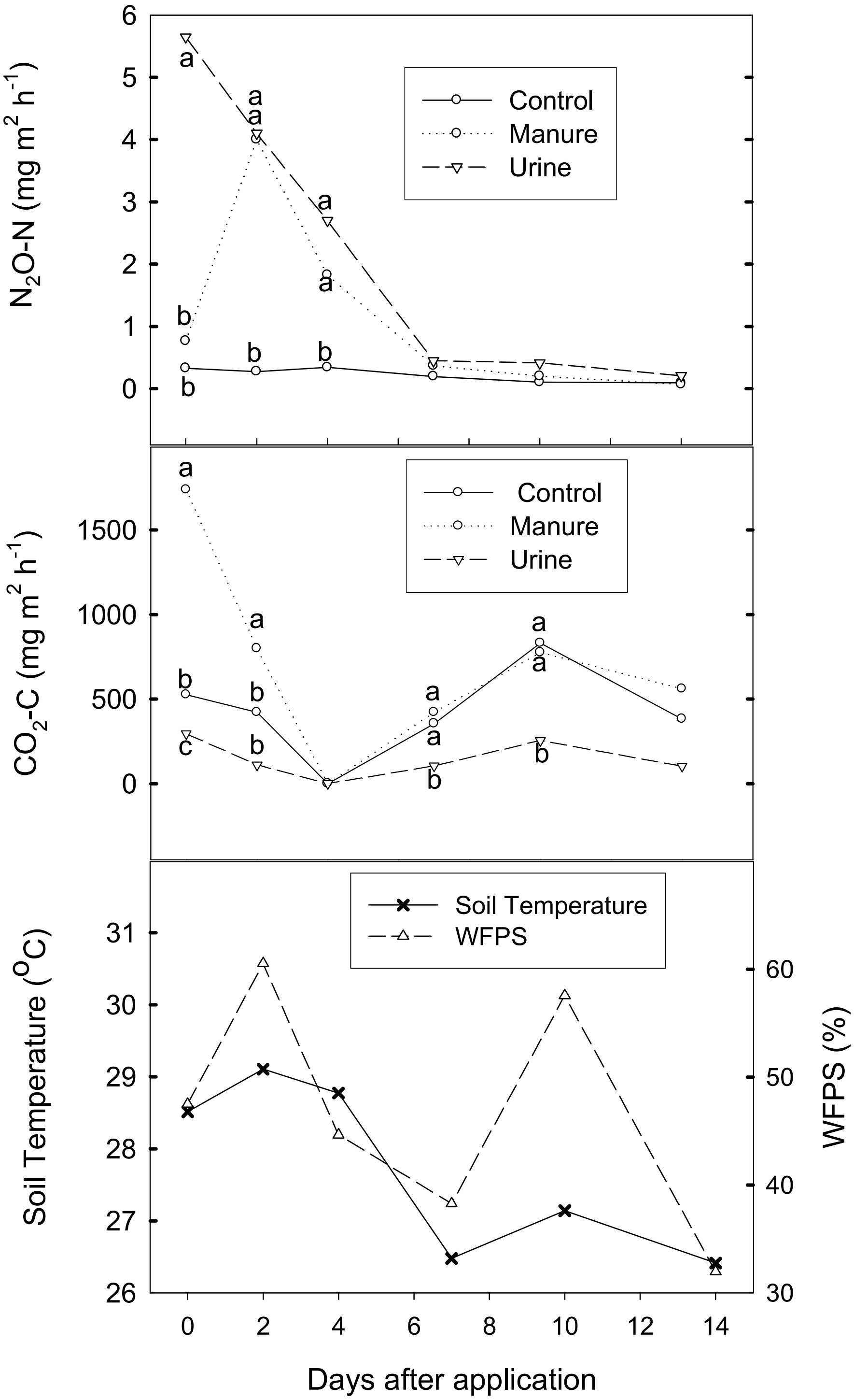

Fig. 4.3. Nitrous oxide and CO_2 emissions, soil temperature, and water-filled pore space of the upper 10 cm during the second N_2O study on the Wrens farm.

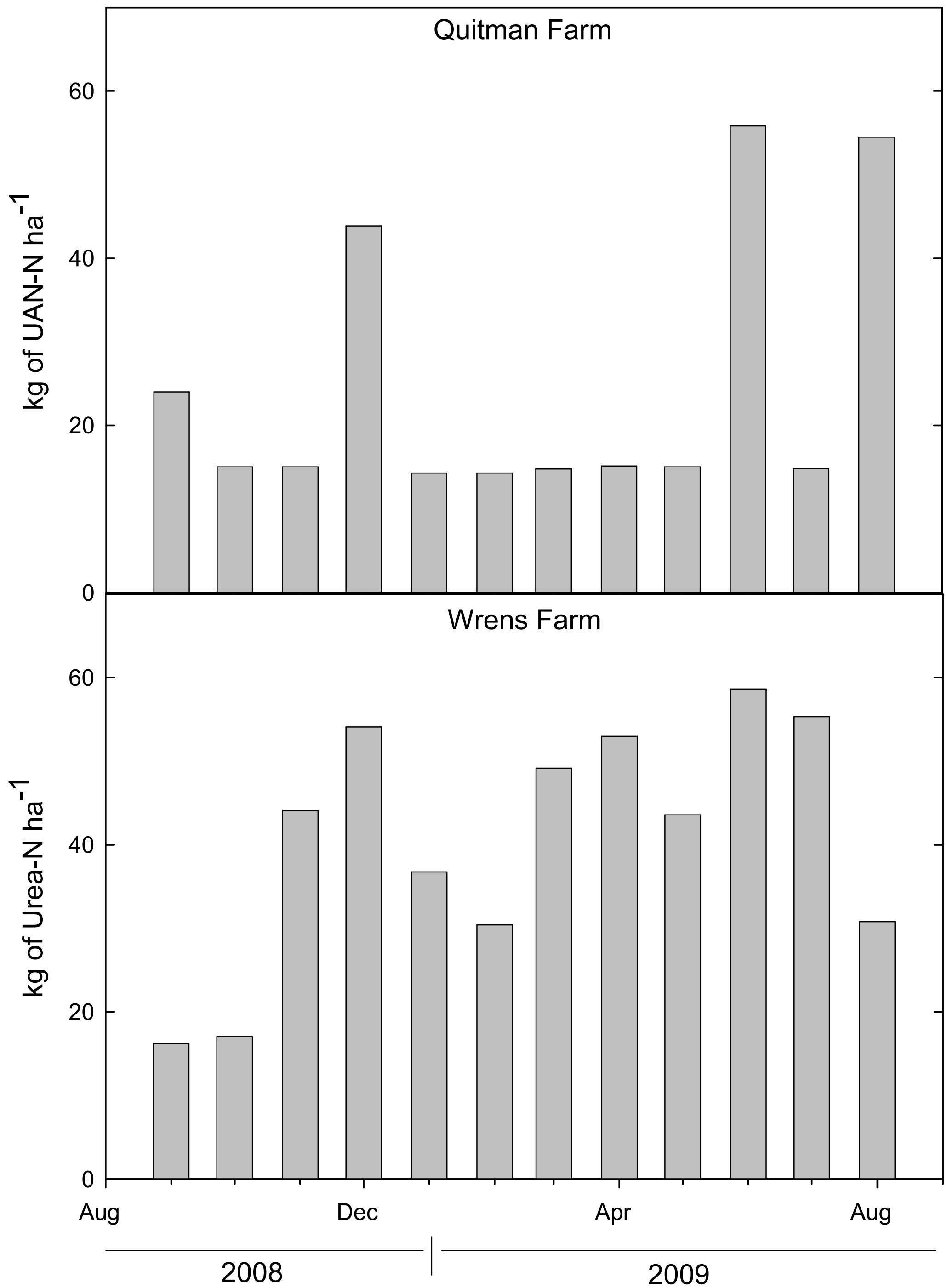

Fig. 4.4. Monthly amounts of N fertilizer applied on the Quitman and Wrens farms.

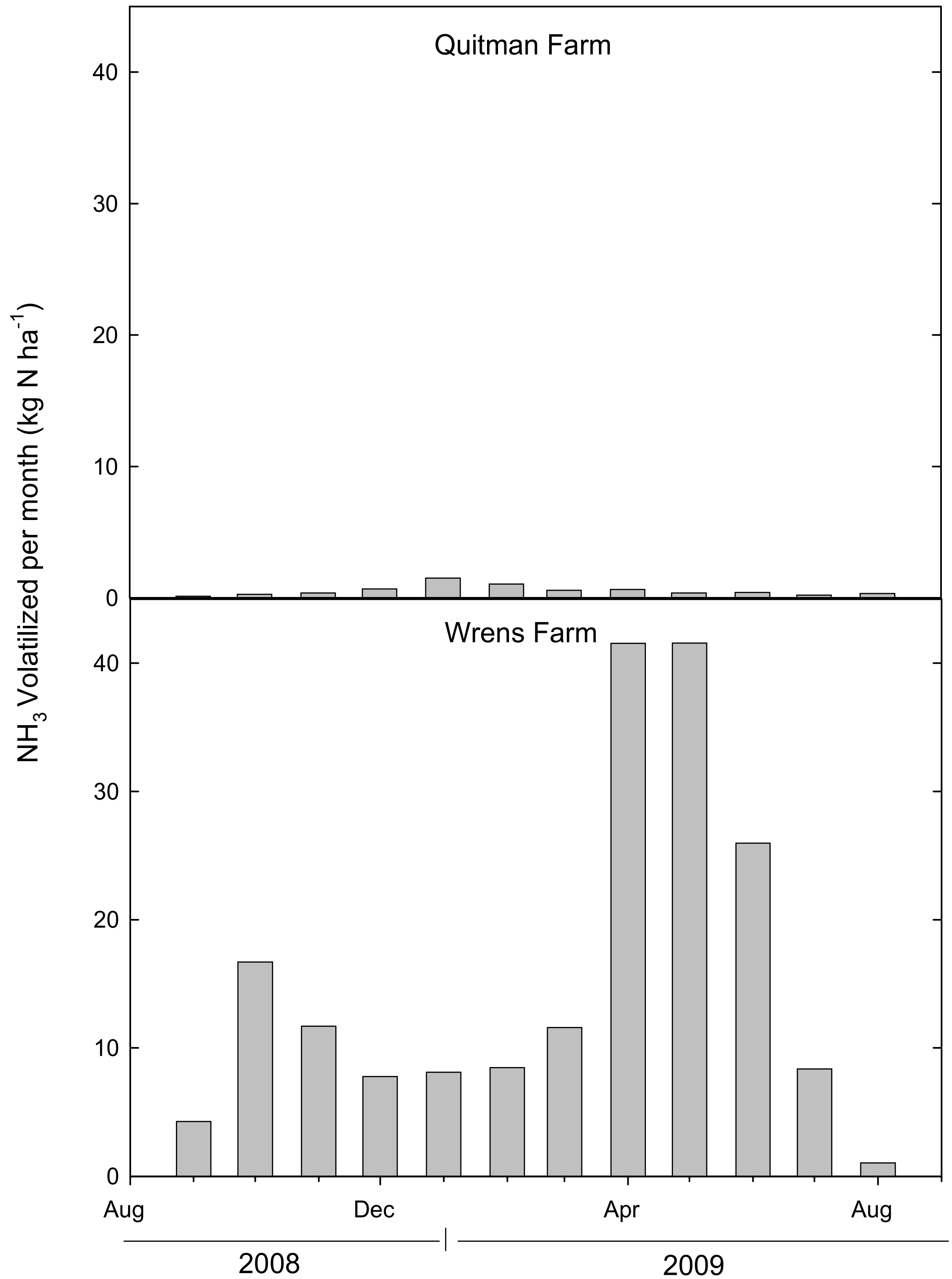

Fig. 4.5. Monthly amounts of NH_3 volatilized on the Quitman and Wrens farms.

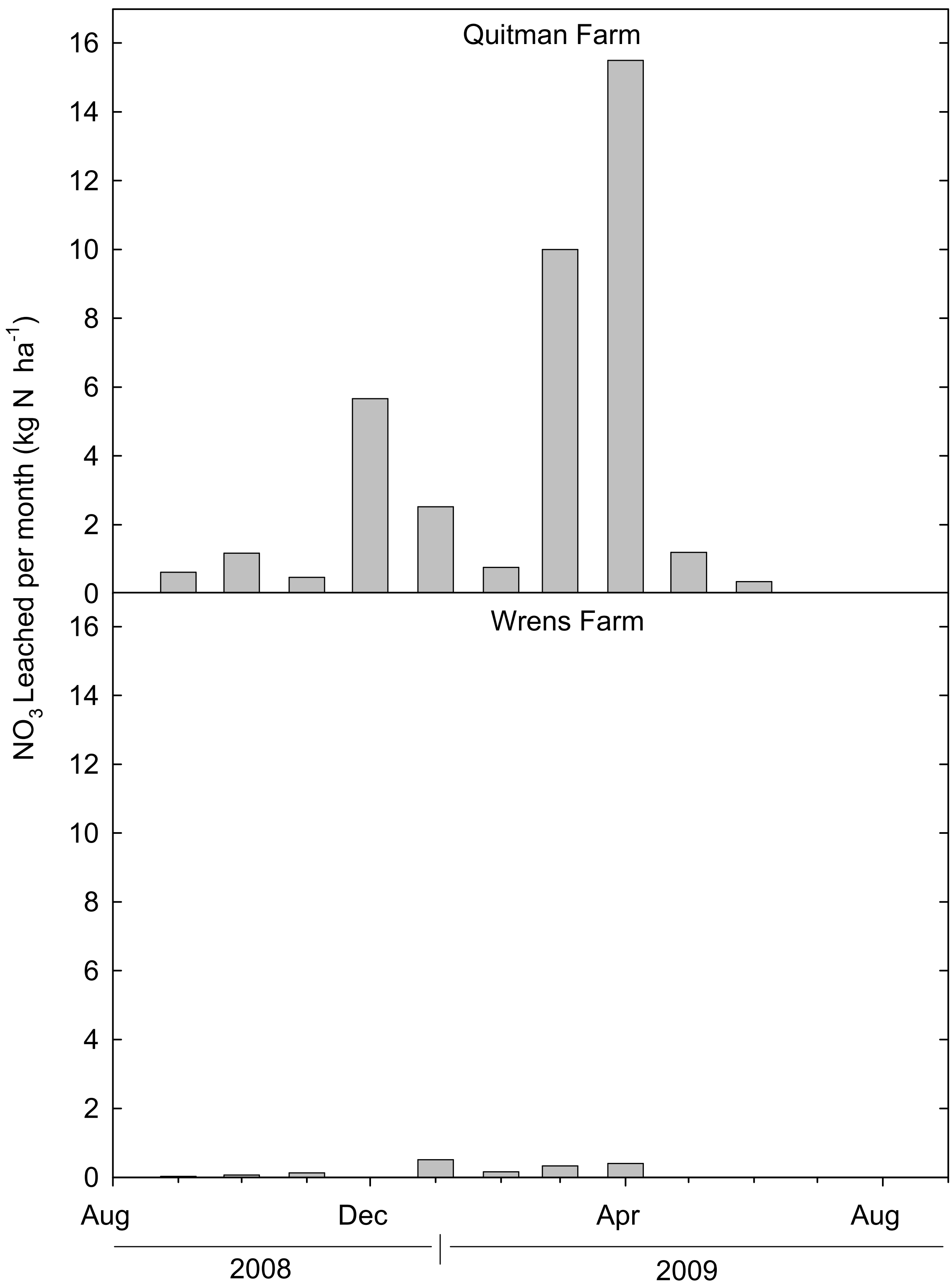

Fig. 4.6. Monthly amounts of NO_3 leached on the Quitman and Wrens farms.

CHAPTER 5

SUMMARY AND CONCLUSIONS

Management intensive pasture-based dairies are now being viewed as an alternative to traditional confinement systems. Grass-based dairies have lower operating costs, better animal health, and potentially less environmental impact. The objectives of this research were to 1) estimate seasonal N losses through ammonia volatilization and nitrate leaching for two irrigated pasture-based dairies (Quitman and Wrens farms) located in the Coastal Plains of Georgia, and 2) develop a farm-gate, N balance for each farm. Overall ammonia losses amounted to 32% of the applied N at the Wrens farm and 2% of the applied N at the Quitman farm. Large ammonia losses at the Wrens Farm were attributed to the use of granular urea applied on the soil surface without sufficient immediate irrigation to incorporate the fertilizer. Ammonia losses at the Quitman farm were relatively small because of the use of UAN through the irrigation system, which presumably allowed a deeper movement of the fertilizer into the soil. Nitrate losses corresponded to 0.3% of applied N at the Wrens farm and 12.5% of applied N at the Quitman farm. The larger losses at the Quitman farm were due to larger water fluxes at the 1-m depth coupled with larger nitrate concentrations in the leachate. The N balance developed for each farm showed that although total N inputs into each farm were similar, the sources were different. Nitrogen fertilizer represented 79% of total N inputs at the Wrens farm, but only 41% of N inputs at the Quitman farm. In contrast, grain feed N as percentage of total N inputs was 16% for the Wrens farm and 53% for the Quitman farm. Nitrogen in milk exports was similar in both farms (16 to 17% of N inputs). As a result of the large differences in NH_3 losses between farms, surplus N (Inputs-Outputs) was greater at the Quitman (77% of N inputs) than at the Wrens farm (52% of N inputs). Most of the surplus N was probably immobilized in the soil, or present in the form of undecomposed fecal matter. These results show that environmental and soil conditions as well as management practices play important roles in the transfer of N on these intensively grazed systems. Additional research should be carried out on the individual processes of N transfer in these systems to better identify the forms in which surplus N is present.