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Metrics for Nitrate Contamination of Ground Water at CAFO Land Application Sites -
Arkansas Dairy Study

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Arkansas Dairy Study

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Notice

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Foreword

The U.S. Environmental Protection Agency is charged by Congress with protecting the Nation's land, air, and water resources. Under a mandate of national environmental laws, the Agency strives to formulate and implement actions leading to a compatible balance between human activities and the ability of natural systems to support and nurture life. To meet this mandate, EPA's research program is providing data and technical support for solving environmental problems today and building a science knowledge base necessary to manage our ecological resources wisely, understand how pollutants affect our health, and prevent or reduce environmental risks in the future.

The National Risk Management Research Laboratory is the Agency's center for investigation of technological and management approaches for preventing and reducing risks from pollution that threatens human health and the environment. The focus of the Laboratory's research program is on methods and their cost-effectiveness for prevention and control of pollution to air, land, water, and subsurface resources; protection of water quality in public water systems; remediation of contaminated sites, sediments and ground water; prevention and control of indoor air pollution; and restoration of ecosystems. NRMRL collaborates with both public and private sector partners to foster technologies that reduce the cost of compliance and to anticipate emerging problems. NRMRL's research provides solutions to environmental problems by: developing and promoting technologies that protect and improve the environment; advancing scientific and engineering information to support regulatory and policy decisions; and providing the technical support and information transfer to ensure implementation of environmental regulations and strategies at the national, state, and community levels.

This work was supported by EPA's Office of Research and Development through the Regional Applied Research Effort (RARE) Program. This program is designed to 1) provide the Regions with near-term research on high priority, Region-specific technical needs, 2) improve collaboration between Regions and ORD laboratories, 3) build the foundation for future scientific interaction, and 4) develop useful tools for state, local and tribal governments to address near-term environmental issues. EPA Region 6 and ORD's Ground Water & Ecosystems Restoration Division (GWERD) recognized the need to evaluate whether properly-designed Comprehensive Nutrient Management Plans (CNMPs) developed for land application of waste from Concentrated Animal Feeding Operations (CAFOs) are truly protective of ground water quality. Funding ($130K total) was awarded to EPA Region 6 (Nancy Dorsey, EPA Region 6 Contact) and administered through GWERD (Elise Striz, Stephen Hutchins, Project Officers), and was used by USDA's Agricultural Research Service (David Brauer, USDA-ARS Contact) to conduct two separate site investigations at CAFO facilities where CNMPs were being followed. The objective was to conduct comprehensive sampling of soil, soil water, and crops for nutrients throughout the growing season to determine which simple soil/crop metrics are the best indicators of the potential for nutrients to escape the root zone and become a threat to ground water. This report describes the site investigation conducted by Dr. Philip A. Moore, Jr., and Dr. David Brauer for a dairy farm in Arkansas. The other site investigation was conducted by Dr. Jerry L. Hatfield for a swine operation in Iowa and is described in the companion report.

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Diagrams and Figures

Abbreviations

CAFOs Concentrated Animal Feeding Operations

HUA   High Use Area

LSD   Least Significant Difference

MCL   Maximum Contaminant Level

N   Nitrogen

NH_4-N   Ammonium Nitrogen

NMP   Nutrient Management Plan

NO_3-N   Nitrate Nitrogen

P   Phosphorus

Acknowledgements

Executive Summary

Nitrate is the most common chemical contaminant found in ground water. Recent research by U.S. EPA has shown that land application of manure can cause nitrate contamination of ground water above the maximum contaminant levels (MCLs) of 10 mg NO_3-N/L at significant depths. This finding and similar ones across the nation are raising concerns about the potential for manure to degrade ground water quality near concentrated animal feeding operations (CAFOs). The objectives of this research were to determine if nutrient management plans (NMPs) for CAFOs are inherently protective of ground water and which metrics can be used as red flags to identify when the land application practices pose a risk to ground water.

A study was conducted for one year (August 2004 to July 2005) on a typical dairy farm in northwest Arkansas. The dairy had 250 cows and utilized 27 ha (67 acres) for manure application. Effluent from the holding pond was sprayed onto four 4.05 ha (10-acre) fields each year. According to the farmer's NMP, effluent applications were to occur during the growing season after the soil had dried considerably, which did not occur until August in 2004. Four small (10 x 10 m) plots were established in each of two of these spray fields. Stainless steel lysimeters were installed to a depth of 1 m and sampled weekly. Three soil cores were taken periodically from seven depths (0-5, 5-10, 10-20, 20-40, 40-60, 60-80 and 80-100 cm). Soil samples were collected 20 times throughout the year. These samples were analyzed for soluble components as well as exchangeable ammonium and Mehlich III extractable P. Plant samples were also analyzed for nitrate.

Nitrate levels in lysimeter samples were high, with peaks in excess of 100 mg NO_3-N/L. The amount of N applied via effluent averaged 280 kg N/ha (250 lbs N/acre), which was not believed to be sufficient to cause such high levels. Lysimeter P concentrations were also very high. Beginning in November, it was observed that the farmer utilized the 4.05 ha fields as a loafing area or high use area (HUA) for his cows. We estimate that at the observed stocking rate (31 cows/ha) as much as 840 kg N/ha (750 lbs N/acre) was being added to these fields via direct waste deposits from the cows. When coupled with the effluent application, the total N loading to these fields was approximately 1100 kg N/ha (1000 lbs N/acre) in one year.

When state officials were contacted to determine if direct deposits were taken into account when determining maximum N application rates for farms spreading liquid manures in Arkansas, we discovered that these deposits by the cows were not accounted for. Currently, the Arkansas Phosphorus Index is being revised. During this revision we will attempt to change regulations so that growers cannot apply effluent to a HUA unless the direct deposits are properly accounted for. The best predictor of high nitrate in the lysimeter samples was leaf tissue nitrate concentration with R-square of 0.82. Since nitrate toxicity in cows is a problem that will negatively affect production, dairy farmers could easily be convinced to monitor this parameter. High nitrate levels in forage would allow them to know which fields were receiving too much N, and allow them to alter applications accordingly.

1.0
Introduction

Animal agriculture is important to the economy of both Arkansas and the nation. Modern farms often have large numbers of animals and a relatively limited land base to apply the manure. This can lead to the problem of over application of nutrients, particularly nitrogen (N) and phosphorus (P) to agricultural lands.

Nitrate (NO_3^-) is soluble in water, hence, it can be easily leached from soils into ground water. As a result, nitrate is the most ubiquitous chemical contaminant in the world's ground-water supplies (Spalding and Exner, 1993). High concentrations of nitrate in drinking water can cause problems if consumed by infants possibly resulting in a serious illness called methemoglobinemia or "blue-baby disease." Blue-baby disease is due to conversion of nitrate to nitrite by the immature gastrointestinal tract, which results in low oxygen levels in the blood.

As a result of this potential human health threat, the U.S. EPA established a maximum contaminant level (MCL) of 10 mg nitrate-nitrogen (NO_3-N)/L for nitrate in drinking water (U.S. EPA, 1995). Unfortunately, Nolan et al. (1998) reported that U.S. EPA MCL was exceeded 24 percent of the time in the United States when data from 1400 wells sampled between 1993 and 1995 were analyzed. Approximately 20 percent of the population of France (10 million people) depends on ground water with nitrate levels above the European Community's upper threshold of 11.3 mg NO_3-N/L (Spalding and Exner, 1993). Nitrate contamination of ground water near intensive vegetable production has been reported in Japan (Babiker et al., 2004).

Several surveys of ground-water nitrate levels have been completed in the past in areas considered prone to high nitrate, including areas with high poultry production, like Arkansas (Arkansas CES, 1990; Steele and McCalister, 1991) and Delaware (Ritter and Chirnside, 1984). The number of wells with nitrate levels greater than 10 mg NO_3-N/L was relatively low in Arkansas (Steele and McCalister, 1991). This was not the case in Delaware, particularly in the number one broiler producing county (Sussex County) where 37 percent of the wells had nitrate levels above the MCL (Ritter and Chirnside, 1984). Ritter and Chirnside (1984) concluded that nitrate leaching from poultry manure was likely the major source of the high ground-water nitrate levels in this county. Only 3.2 percent of the 1232 wells sampled in a ten county area of Arkansas had nitrate levels above the MCL and most of these were in the poultry producing areas (Arkansas CES, 1990). However, Steele and McCalister (1991) reported the average nitrate concentration was only 3 mg NO_3-N/L in areas receiving heavy applications of poultry litter in western Arkansas.

Adams et al. (1994) studied nitrate leaching in a Captina silt loam soil in northwest Arkansas for 1 year after being fertilized with various rates (0, 5, 10 and 20 Mg/ha or approximately 2.5 to 10 tons/acre) of poultry litter and laying hen manure, which corresponded to N application rates of 0, 220, 440 and 880 kg N/ha. Lysimeter data taken from 1.2 m depth showed that only the 10 and 20 Mg/ha rates resulted in soil solution nitrate values in excess of 10 mg NO_3-N/L with maximum concentrations of 24 and 37 mg NO_3-N/L, respectively. Adams et al. (1994) concluded that if manure application rates are made at the recommended rates in Arkansas, less than 11 Mg/ha or 5 tons/acre (Daniels et al., 2008), then excessive nitrate leaching should not occur. These results confirmed those of Marriott and Bartlett (1975), who showed that manure application rate played an important role in nitrate leaching. They indicated that manure could be applied at twice the crop's N needs with minimal threat of nitrate leaching to the ground water, provided the manure is applied to a deep well-drained soil and the crop is harvested.

Although rate of N application is an important determinant of the leaching potential of nitrates, other factors also influence the concentration of nitrates in soil water below the plants' rooting depth. Both the amount of and the timing of precipitation events and irrigation water applications relative to time of N applications affect the amount of nitrate found deep in the soil profile (Gärdenäs et al., 2005; van Es et al., 2006). There may be a value in the use of estimates of crop evapotranspiration to scheduling irrigation amounts and frequency to limit the amount of nitrate leaching through the soil profile (Gärdenäs et al., 2005). Leaching potential of nitrate through coarse texture soils is greater than for finer texture soils (van Es et al., 2006). Source of the N, including the type of manure, also influences the rate at which nitrate moves through the soil profile (Giullard and Kopp, 2004; Wu and Powell, 2007).

Spalding and Exner (1993) stated that high temperatures, abundant rainfall and relatively high organic contents in Coastal Plain soils of the southeastern United States promote denitrification below the root zone and naturally remediate nitrate leaching into ground water. In North Carolina, Gilliam (1991) found that high levels of

nitrates (15-20 mg NO_3-N/L) occurred in soil solutions in Coastal Plain soils cropped to corn. However, these high concentrations were not measured below 4 m. Gilliam (1991) attributed these low nitrate levels at greater depths to denitrification (soluble organic carbon compounds provide an energy source for microbial reduction of nitrate).

Several methods for monitoring nitrate leaching have been used in the past. Nitrate analyses of soil cores provided estimates of nitrate leaching as reliable as two different types of soil lysimeters (Zotarelli et al., 2007). Zhu et al. (2002) and Toth et al. (2006) demonstrated that passive capillary lysimeters also provided reliable estimates of nitrate leaching, while being easier to install and maintain than other types of lysimeters. The problem with these methods is that they are labor intensive during installation and/or sample collection, thus limiting their use by farmers.

Recent work by U.S. EPA personnel in Oklahoma have demonstrated that land application of swine manure can cause nitrate contamination of soil water above the MCL at depths greater than 10 m (Elise Striz, unpublished data). These findings, along with similar findings around the country, are raising concerns for soil and ground-water degradation on or adjacent to concentrated animal feeding operations (CAFOs). Currently, land application of manures from CAFOs must follow a nutrient management plan (NMP) in most states. In Arkansas, NMPs for liquid manures have been regulated for over a decade by Regulation 5, which states that manure applications will be based on the N needs of the crop. One of the main underlying assumptions of using a well designed and executed NMP is that ground water will be protected from excessive amounts of nitrate or other nutrients.

In the southern United States, attention has focused on the potential of animal manure from poultry and integrated poultry and beef cattle operations contributing to ground-water contamination because of the number of these operations. In 1997, poultry and beef cattle production comprised 50 percent of the agricultural income in Arkansas and over 90 percent of the animal manure production (VanDevender, 1997). Dairy farms in Arkansas, on the other hand, accounted for 2 percent of the farm income and 5 percent of the animal manure production. The stocking rate of cattle (number of head/ha or acre) tends to be greater with dairy than beef cattle. Dairy cattle routinely require supplemental feeding with both energy- and protein-rich feeds, thus leading to an importation of nutrients to the farm. A disproportionate amount of these nutrients are fed to dairy cattle during that period of the year when forages are dormant. The potential for nutrient run-off or leaching is greater when cattle are being fed

on dormant grasslands (Owens and Shipitalo, 2006). The amount of water soluble P in dairy manure is relatively high (Shigaki et al., 2006). Several studies have indicated that the amount of soluble P applied as manure is directly associated with the potential for P losses in run-off (DeLaune et al., 2004a; 2004b). Soupir et al. (2006) indicated that nutrients are more readily lost from direct deposition of dairy manure onto the soil than losses from other application methods. These previous findings indicate that there is the potential for ground-water contamination on or adjacent to southern dairy operations, especially when rainfall exceeds evapotranspiration and soils are permeable to water.

The objectives of this research were to determine if an NMP, when properly executed, consistently protects ground water, and, if not, to determine what soil/crop/manure metrics could be utilized to predict potential ground water degradation.

Materials and Methods

Site Selection

The site selected for this study was a dairy farm located in Washington County in northwest Arkansas. The initial meeting with the landowner was made on July 20, 2004. The dairy had approximately 250 cows on 27.1 ha (67 acres, 47 acres were grazed; 20 acres were hayed). Lagoon effluent from the farm was applied to four 4.05 ha (10-acre) fields of bermudagrass (*Cynodon dactylon* (L.) Pers). Two of the four fields, designated as Field 1 or 2, were randomly chosen for the study. Within each of the two fields, four plots (10 x 10 m) were established (Diagram 1). According to the Washington County Soil Survey (USDA, 1969), the soil type in both fields was exclusively a Jay silt loam (fine-silty, mixed, active, thermic Oxyaquic Fragiudalf) on 1-3 percent slope. Characteristics of soil recovered from coring to 1 m depth were consistent with this determination (data not shown). According to the Washington County Soil Survey (USDA, 1969) the Jay series consists of well-drained, slowly permeable soils that have a subsoil fragipan between 0.4 and 1.0 m of the soil surface. A full description of the Jay silt loam is available (NRCS, 2000). No attempt was made to determine the depth at which the fragipan occurred. The landowner was able to utilize these fields according to his management scheme, thus cattle were not excluded from the plots. Rainfall data were obtained from a USGS gauge station located near the fields (USGS, 2009).
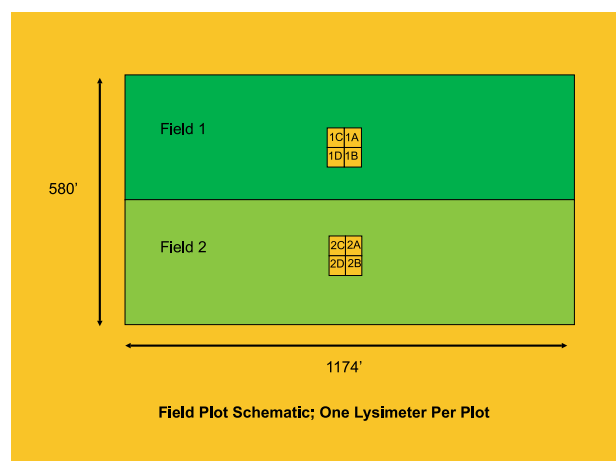

Diagram 1. Schematic showing location of plots within Fields 1 and 2.

Soil Samples

Three soil cores (5 cm in diameter) were collected from each plot on each sampling date. The cores were divided into 7 discrete samples by depth (0-5, 5-10, 10-20, 20-40, 40-60, 60-80 and 80-100 cm). Soils from each depth of the three cores were combined for each plot. Thus, there were a total of 56 soil samples (8 plots x 7 depths) taken at each sampling time. First soil samples were taken on July 24, 2004. Soil samples were collected 19 additional times over the next year, with the bulk of the samples being collected just after dairy effluent applications. Collection of soil samples terminated prior to effluent application in 2005.

Soluble reactive phosphorus, nitrate-N, and ammonium-N were extracted using a 1:10 (soil:water) extraction for one hour (Self-Davis et al., 2000). A sequential KCl extraction was conducted on the soil for exchangeable ammonium. Nitrate (+nitrite) was determined using the Cd reduction method on filtered (0.45 m) samples (Method 418-F; APHA, 1992). Soluble reactive P samples were also filtered (0.45 m), and acidified to pH 2.0 with HCl. Soluble reactive P was determined using the ascorbic acid technique (APHA Method 424-G; APHA, 1992). Ammonium was determined with the salicylate-nitroprusside technique on filtered (0.45 m), acidified samples (Method 351.2; U.S. EPA, 1979). Soil samples were also analyzed for Mehlich III P (Mehlich, 1984). Levels of soil constituents were expressed as mg per kg of dry soil. These concentrations were converted to pounds per acre assuming 2,000,000 pounds of soil per acre-foot of topsoil.

Lysimeter Samples

One stainless steel lysimeter (50 cm long and 5 cm in diameter) was installed within each of the four 10 x 10 m^2 plots in each field on August 2, 2004 (Diagram 2). Both the equipment and installation were essentially as described by Adams et al. (1994). Installation began by digging a hole of sufficient size to accommodate an armor valve box (approximately 15 cm in diameter and 20 cm in height). From the bottom of this hole a second hole was dug at a 45-degree angle to the soil surface, in which the stainless steel lysimeter and the attached vacuum tubes were placed. Excavated topsoil and subsoil were kept separately. After placement of the lysimeter, the hole was filled with the removed material starting with excavated subsoil. Such an installation minimizes the disturbance
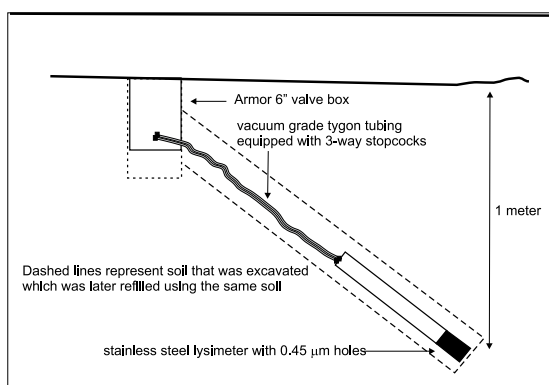
Diagram 2. Stainless steel lysimeter installation.

to the soil immediately above the sample collection chamber. Leachate from the sample collection chamber was collected by the application of a vacuum. Attempts to gather ground-water samples from the lysimeters were made on a weekly basis. Between September 9, 2004 and October 31, 2004, ground-water samples were obtained from less than half of the lysimeters. During November 2004, ground-water samples were obtained from over half of the lysimeters, but not all. From December 1, 2004 through May 14, 2005, ground-water samples were obtained from all lysimeters. After May 14, 2005, the number of lysimeters with ground-water samples declined until only two lysimeters in Field 2 had samples. Attempts to gather ground-water samples were terminated prior to effluent application in 2005. Chemicals present in the soil water at a depth of 1 m are likely to continue to percolate downward until reaching ground water because a vast majority (over 95 percent) of roots and associated organic C are located at a soil depth of less than 0.9 m in bermudagrass dominated agro-ecosystems (Carreker et al. 1977; Franzluebbers and Stuedemann, 2005). Lysimeter samples were analyzed for pH, electrical conductivity (EC), total organic C (TOC), total nitrogen (TN), ammonium (NH_4^+), nitrate (NO_3^-), soluble reactive phosphorus (SRP), and total P (TP). Ammonium was determined with the salicylate-nitroprusside technique (Method 351.2; U.S. EPA, 1979). Soluble reactive P samples were filtered (0.45 m), acidified to pH 2.0 with HCl and frozen. Soluble reactive P was determined using the ascorbic acid technique (Method 424-G; APHA, 1992). Total N and total organic C were analyzed using a Skalar total N and C analyzer. Total P and metals were determined using a Spectro inductively coupled argon plasma emission spectrometer following acid digestion.

Effluent Application and Collection of Applied Effluent Samples

Effluent applications were made according to an approved NMP. Application amounts were based on N needs of the forage. Liquid manure from the holding pond was applied via a travel gun on Field 2 on August 3, 2004 and on Field 1 on August 18, 2004. Length of effluent application was 1 hr 45 min for Field 1 and 2 hr for Field 2. The pressure at which the liquid was sprayed was 207 kPa (30 pounds per square inch) on both fields. Four shallow plastic pans (16.5 x 30.5 cm) were placed in each of the four plots during effluent application. The effluent collection area represented approximately 2 percent of the plot area. The volume of effluent captured in each was recorded. The captured effluent was analyzed for total N and nitrate concentrations. The captured effluent had an average total N content of 651 and 514 mg N/L for Fields 1 and 2, respectively. Nitrate concentrations were 0.30 and 0.38 mg NO_3-N/L, respectively, for effluent collected on Fields 1 and 2. These results indicate that most of the N in the effluent was not in the nitrate form.   The average application rate applied to the plots in Field 1 was 602,100 L/ha (or 64,100 gallons/acre). The amount of effluent captured in the collection pans in the plots in Field 2 was considerably less, averaging 330,000 L/ha (or 35,100 gallons/acre). Using the total N content of the collected effluent, N application rates of 391 and 168 kg N/ha (or 349 and 151 lbs N/acre) were calculated for Fields 1 and 2, respectively. The average total N application rate for both fields was 280 kg N/ha (or 250 lbs N/acre).

What was captured in the pans was probably representative of what the plots received, but may not have been representative of what was applied to the entire field. The traveling gun was spraying effluent on Field 2 for a slightly longer period of time than on Field 1, 2 hours versus 1.75 hours (see above). Hence, Field 2 should have had slightly more manure applied than Field 1. The reason for discrepancy between what was applied to the field and what fell on the plots is probably the non-uniform application which was made by the gun. The location of the plots, direction and speed of wind, etc., relative to the position of the traveling gun in the field were not the same. Very few, if any, growers spreading liquid manure in Arkansas have center pivot irrigation systems. They have traveling guns, like the one this grower utilized. Hence, uneven applications within fields are probably typical of on-farm situations. No weather station was installed at the experimental site. Daily precipitation amounts were obtained from the USGS stream gauge 07048600, which was located within 10 km of the research site (USGS, 2009).

Forage Samples

Leaf tissue was collected from vegetation growing in each of the 8 plots every 30-40 days. Above ground biomass was clipped within 2 cm of the soil surface from 10 randomly selected areas within each plot, each approximately 15 cm in diameter. A sub-sample from the leaf material collected from each plot was dried at 60°C for 48 hours, ground to pass a 40-mesh screen and nitrate-N was determined via nitrate reductase assay (MacKown and Weik, 2004).

Statistical Analyses

Data were analyzed by analysis of variance using SAS (SAS Institute, 1985). The experimental unit was a plot within one of the two fields. The experimental design included sampling dates, plots within field and fields as the main factors. In the statistical analyses, missing data from a sampling date were treated as missing values. Least significant differences (LSD) at $P = 0.05$ are included in figures and text for mean comparison tests. Days mentioned in text and accompanying figures refer to days after the study was initiated, i.e. July 27, 2004.

3.0
Results and Discussion

Precipitation Data

Daily precipitation from June 1, 2004 to September 1, 2005 is presented in Figure 1. Sufficient rainfall occurred in June and July 2004 (Day -56 to Day 0) to prevent the application of effluent to the bermudagrass fields until early August, as to comply with the soil moisture parameters of the farm's NMP. A substantial rainfall event occurred immediately after the effluent applications. Approximately 8-10 cm of water were applied to the fields by effluent application and rainfall between August 1 (Day 5) and September 9, 2004 (Day 44), the date that the first soil water samples were obtained from the lysimeters. Little rainfall fell during the fall of 2004. Several rainfall events occurred around December 1, 2004 (Day 127) and February 1, 2005 (Day 189). Frequent rainfalls of over 1 cm occurred between April 1 and July 1, 2005 (Days 248-339).
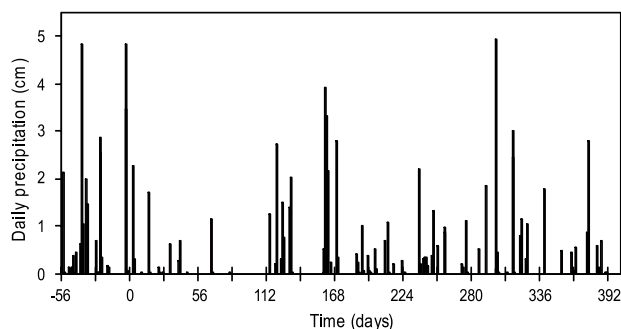

Figure 1.   Daily precipitation between June 1, 2004 (Day -56) and September 1, 2005 (Day 401). Time zero is July 27, 2004. Daily precipitation was obtained from USGS gauge station 07048600 located on the White River near Fayetteville AR. This gauge station is located within 10 km of the experimental site.

Lysimeter Data

The first lysimeter ground-water samples were collected on September 9, 2004, approximately three to four weeks after effluent applications were made. Nitrate levels in the lysimeters were exceedingly high (Figure 2), exceeding 200 mg NO_3-N/L at Day 44 of the study (September 9, 2004). Nitrate levels were above 100 mg NO_3-N/L from Day 0 to Day 150 (mid-December 2004). Nitrate levels steadily decreased until around Day 200 (February 2005), then increased again. The average nitrate concentration in Field 1 was significantly ($P < 0.05$) higher than Field 2 (149 versus

90 mg NO_3-N/L). This may have been due to the higher N application rate in the plots in Field 1 versus Field 2. Ground-water samples were collected from at least one lysimeter in Field 2 for a longer period of time compared to Field 1, thus accounting for the data beyond Day 320 for Field 2. Trends in total N levels in lysimeter samples were similar to those of nitrate (Figure 3).
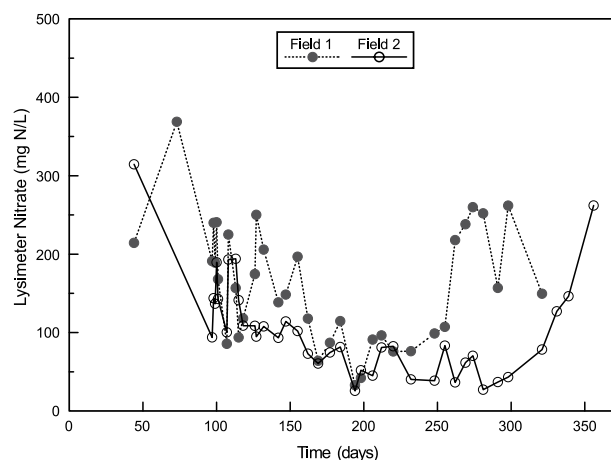

Figure 2.   Lysimeter nitrate concentrations as a function of time. Time zero is July 27, 2004, date of the first soil sampling. Days 100, 200 and 300 correspond to November 4, 2004, February 12, 2005 and May 23, 2005, respectively. Effluent applications occurred on August 3 and 18, 2004 for Fields 2 and 1, respectively, or Day 6 and 21, respectively. Data are means across four plots (n = 4) within each field at a sampling date (LSD = 22.0 mg N/L).

The discovery of higher than expected nitrate values at the beginning of the study was hard to explain at first, since N applications from effluent appeared to be insufficient to cause such elevated levels. As stated earlier, Adams et al. (1994) only found a high of 37 mg NO_3-N/L in a soil fertilized with manure containing 880 kg N/ha; an N application rate far in excess of what was applied to these fields. Moore et al. (2000) measured much lower nitrate levels in soil solutions on two typical poultry/beef farms in northwest Arkansas during a two year study, with the highest observed nitrate concentrations being 2.8 mg NO_3-N/L, roughly 100 fold lower than the peak values found in this study.
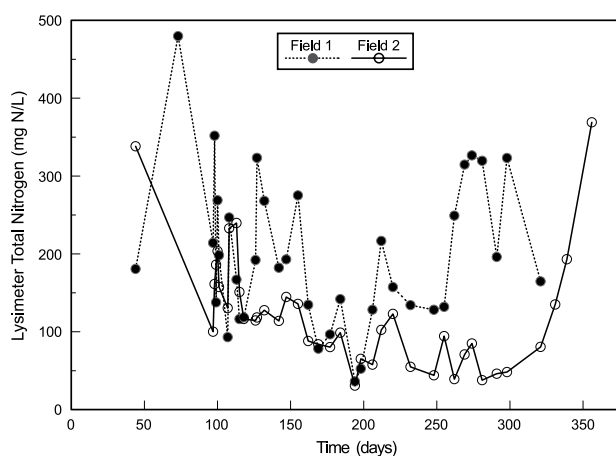
Figure 3. Concentrations of total N (TN) in lysimeter ground-water samples as a function of time. Time zero is July 27, 2004, date of the first soil sampling. Days 100, 200 and 300 correspond to November 4, 2004, February 12, 2005 and May 23, 2005, respectively. Effluent applications occurred on August 3 and 18, 2004 for Fields 2 and 1, respectively. Data are means across four plots within each field (n = 4) at a sampling date (LSD = 29.4 mg/L).

Beginning in early November 2004, it was observed that the dairy farmer was using the two fields where effluent was applied for a loafing area for the cattle. Within a month of using this area for this purpose, the soil surface was covered in a layer of manure that was 1-2 cm deep (direct deposits from the dairy cows). The assumption was made that using the effluent application fields for loafing areas was a direct violation of the nutrient management plan. However, personnel from USDA/NRCS and the University of Arkansas indicated that no such restrictions were in place and they routinely did not account for direct deposits of N and P when evaluating or developing NMP for effluent applications.

The grower at this farm was using the two 4.05 ha (10-acre) spray fields as the dairy loafing area in the winter for all of his cows. Hence, 250 cows were being kept on roughly 8.1 ha (20 acres) all day, except when the cows were moved to and from the milking parlor. Thus, the stocking rate was about 31 cows/ha (or 12.5 cows/acre). A typical dairy cow weighing 500 kg (1,100 lbs) excretes roughly 0.23 kg N/day (0.5 lbs N/day) in manure and urine (USDA, 1992). Therefore, approximately 7 kg N/ha (6 lbs N/acre) were being added each day by the cattle via direct deposits. These fields were used as a loafing area for 3 to 4 months during the year of the study. Responses from the farmer indicate that use of these fields as a loafing area for 3 to 4 months annually had been a common practice for several years. If the fields were used as a loafing area for 90 to 120 days, then the

total direct deposit of N from cattle would be 620 to 820 kg N/ha (560 to 750 lbs N/acre). These estimates indicate that direct deposit of manures added three times the average amount of N that was added via effluent applications (i.e. 280 kg N/ha or 250 lbs N/acre). Thus, the total annual N loading on these fields was at least 800 kg N/ha and could easily have exceeded 1,000 kg N/ha.

As mentioned earlier, nitrate concentrations in ground water have been shown to be directly influenced by the amount of manure applied (Adams et al., 1994; Marriott and Bartlett, 1975). Results of Adams et al. (1994) indicated that N application rates from poultry litter additions had to exceed 440 kg N/ha for ground-water samples taken at 120 cm to exceed 10 mg NO_3-N/L. Results of Adams et al. (1994) predict that lysimeter NO_3-N concentrations in this study should have been less than 10 mg NO_3-N/L if N was being added only from the effluent applications. Lysimeter NO_3-N concentrations in this study typically exceeded that observed by Adams et al. (1994) suggesting that the N application rate was in excess of the previous study's highest rates, 880 kg N/ha. Therefore, total N input of at least 1,000 kg N/ha seems plausible, based on the observed nitrate-nitrogen concentrations in lysimeter samples.

If most of the N additions to these soils were occurring from direct manure deposits during the winter, why did lysimeter nitrate values decline during this time period (Figure 2)? One possible explanation is that manure deposited on the soil surface during the winter did not start to break down until warmer weather in spring. The dominant N forms in dairy manure were not nitrate. Nitrate is the form of soil N most susceptible to leaching. Thus, significant mineralization of the N in the manure would have to occur to produce nitrate for leaching to be detected. This mineralization could account for the increases in lysimeter nitrate and total N levels beginning between Days 225 and 250 (mid-March 2005) in Field 1 and between Days 275 and 300 (mid-June 2005) in Field 2. Neither of these increases in lysimeter concentrations was due to additional effluent applications. Precipitation amounts during the winter months of 2005 were less than historical means (Figure 1), thus downward movement of nitrate and other N compounds into the soil profile would have been minimal. As spring progressed, warmer temperatures coincided with increases in precipitation. Increased precipitation and temperatures would have promoted both N transformation and downward movement of soluble water N compounds, thus explaining increases in lysimeter N concentrations after Day 250.

Although the total N concentrations in lysimeter samples followed the same patterns as nitrate, the values were

about 25 percent higher (Figure 3). Total N is higher because it includes other forms of N, such as ammonium and organic N forms. Mean total N concentrations for Field 1 were higher than for Field 2 (189 versus 108 mg N/L), as was the case for nitrate, and consistent with the higher N application rate on Field 1. Differences in average lysimeter concentrations between the two fields suggest that the addition of N via effluent application was sufficient to alter ground-water levels of nitrate and total N.

Lysimeter ammonium values were less than those for nitrate (Figure 4). Twice during the experiment, ammonium values from the lysimeters in Field 1 exceeded 4 mg N/L. One event occurred around Day 130 (early December 2004) and the other was observed around Day 230 (mid-March 2005). It is unclear what caused these spikes; they did not coincide with either effluent applications or high amounts of rainfall. The mean ammonium concentrations in soil solutions were significantly higher in Field 1 than Field 2 (4.4 versus 0.6 mg NH_4-N/L). These higher concentrations in the plots in Field 1 may have been due to the higher application rate of effluent, and thus, N.
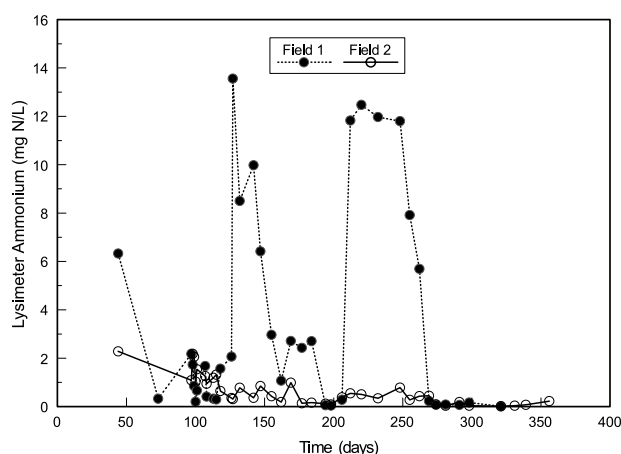

Figure 4.   Ammonium concentrations in lysimeter ground-water samples as a function of time. Time zero is July 27, 2004, date of the first soil sampling. Days 100, 200 and 300 correspond to November 4, 2004, February 12, 2005 and May 23, 2005, respectively. Effluent applications occurred on August 3 and 18, 2004 for Fields 2 and 1, respectively. Data are means across four plots within each field (n = 4) at sampling date (LSD = 1.97 mg/L).

The amount of nitrate in the lysimeters was highly correlated with total N, as was organic N (Figure 5). However, ammonium was poorly correlated to total N concentrations. The slopes from these three relationships indicate that about 77 percent of the total N was nitrate, 22 percent was organic N and 1 percent was ammonium.
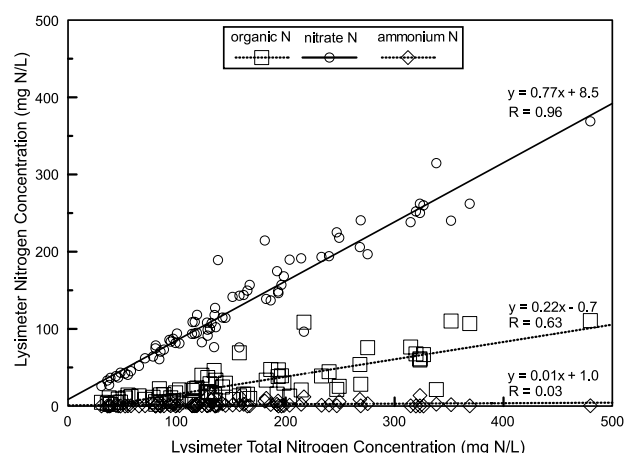

Figure 5.   Relationship between concentrations of nitrate, organic N, and ammonium N to total N concentrations in lysimeter ground-water samples. Regression equations and regression coefficients (R) are presented in figure.

It is somewhat unusual for shallow lysimeters installed in soils in northwest Arkansas to have ground-water samples for a majority of the year as observed in this study. Normally lysimeter samples can only be taken a few weeks of the year following heavy rainfall events. The soils in these two fields were a Jay silt loam, which is known to have a fragipan in the subsoil. The fragipan restricts the downward movement of water, thus leading to a perched water table in the subsoil for at least a portion of the year. Therefore, it is quite possible that the lysimeters were sampling water from this perched water table and that is the reason why lysimeter samples could be collected for a large part of the year.

The chemical composition of the lysimeter samples, which contained high concentrations of nitrate and low levels of ammonium, indicate that the soil solution remained oxidized (under anaerobic conditions ammonia will not be oxidized to nitrate). This is somewhat surprising, since the deeper soil samples were often dark grey to blue in color, which normally indicates anaerobic conditions. However, the average iron and manganese concentrations in soil solution were only 0.21 and 0.14 mg/L, respectively (data not shown), which also indicate oxidized conditions. Had anaerobic conditions been present, then denitrification should have occurred, since plenty of organic C was present. Concentrations of total organic C were typically near 50 mg C/L or above (data not shown).

Soluble reactive P levels in the lysimeters were typically between 0.5 and 2.0 mg P/L, but increased to around 7 mg P/L in Field 1 halfway through the study (Figure 6). Soluble reactive P levels were significantly higher in Field 1 than Field 2 (1.60 versus 1.02 mg P/L). As with total N and nitrate concentrations, the

differences in soluble reactive P between the two fields indicate that the P additions via the effluent application were sufficiently large to affect ground-water sample concentrations. Typically, soil water P values taken from lysimeters at 1 m depth are much lower in northwest Arkansas, ranging between 0.01 to 0.05 mg P/L (Philip Moore, unpublished data). Total P concentrations of water samples taken from the lysimeters were also high (Figure 7). Total P values were only slightly higher than those for soluble reactive P (Figures 6 and 7), indicating that most of the P in the lysimeter samples was soluble reactive P. These data indicate that this soil was saturated with P.

Figure 7. Concentrations of total P in lysimeter ground-water samples as a function of time. Time zero is July 27, 2004, date of the first soil sampling. Days 100, 200 and 300 correspond to November 4, 2004, February 12, 2005 and May 23, 2005, respectively. Effluent applications occurred on August 3 and 18, 2004 for Fields 2 and 1, respectively. Data are means across four plots within each field (n = 4) at a sampling date (LSD = 2.43 mg/L).
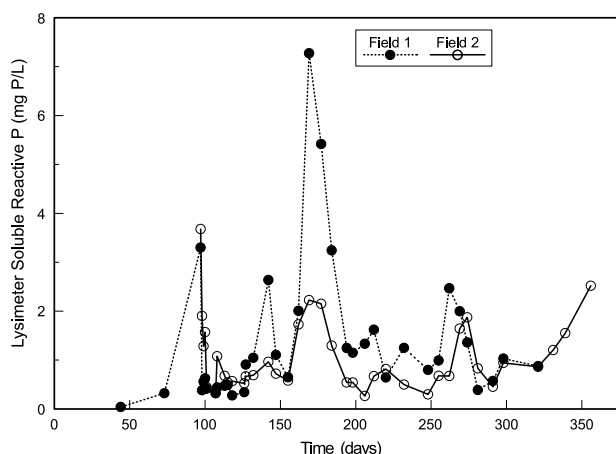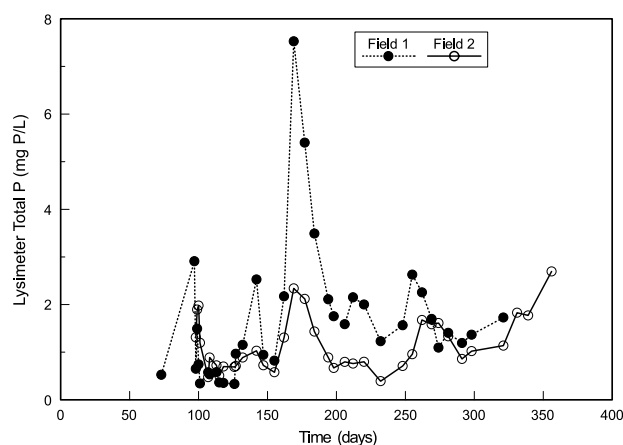

Figure 6. Concentrations of soluble reactive P in lysimeter ground-water samples as a function of time. Time zero is July 27, 2004, date of the first soil sampling. Days 100, 200 and 300 correspond to November 4, 2004, February 12, 2005 and May 23, 2005, respectively. Effluent applications occurred on August 3 and 18, 2004 for Fields 2 and 1, respectively. Data are means across four plots within each field (n = 4) at a sampling date (LSD = 1.97 mg/L).

As with N, a lot of the P present in these fields probably originated from direct deposits. A 500 kg dairy cow excretes roughly 0.03 kg P/day. If the field were used as loafing areas for four months with an average density of 31 cows/ha, then roughly 110 kg P/ha (100 lbs P/acre) would have been deposited. While significant amounts of N may be lost via gaseous emissions, such as ammonia volatilization or possibly denitrification, the only loss mechanisms for P would be runoff or leaching.

Soil Data

One set of soil samples was taken prior to effluent application on Field 2. Three sets of soil samples were taken prior to effluent application on Field 1, and one set was collected the day after effluent application (i.e.,

August 19, 2004). Soil nitrate concentrations in the surface layer (0-5 cm) were high throughout the study (Figure 8). There appeared to be a definite response of soil nitrate concentrations in the surface layer to the effluent application in Field 1. Soil nitrate levels at the soil surface (0-5 cm) in Field 1 varied between 60 and 100 mg NO_3-N/kg soil over the first four sampling dates, three prior to effluent application and one day after. Soil nitrate levels then increased over two-fold, exceeding 150 mg NO_3-N/kg soil, during September and October 2004 (Days 30-100). This peak in soil nitrate levels appeared to occur in response to precipitation that occurred during September and October. Warm temperatures and adequate moisture are necessary for soil microbes to convert the N in the effluent to nitrate. Such a trend was less apparent in Field 2. Soil nitrate levels in the soil surface (0-5 cm) in Field 2 averaged almost 200 mg NO_3-N/kg at the beginning of the study, and increased slightly in the samples on the second sampling date, which was 2 days after effluent application. Surface concentrations of soil nitrate decreased to approximately 80 mg NO_3-N/kg soil by the fourth sampling date (Day 23). As with Field 1, soil nitrate concentrations in the surface layer (0-5 cm) increased between the fourth and fifth sampling dates and remained at an elevated level to approximately Day 100 (Figure 8). These results are consistent with a higher rate of N addition via the effluent to Field 1 compared to Field 2. Soil inorganic N (nitrate plus ammonium) followed similar trends, but peaked between 250 and 300 mg N/kg (Figure 9). Comparisons of the concentration of soil nitrate relative to inorganic

N indicate that nitrate was the dominant inorganic N species in the soil, as was found in the lysimeter samples.
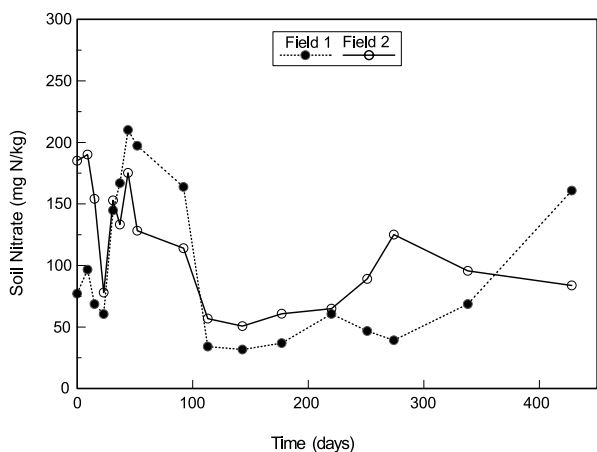

Figure 8. Soil nitrate concentrations in samples taken from 0-5 cm depth as a function of time. Time zero is July 27, 2004, date of the first soil sampling. Days 100, 200 and 300 correspond to November 4, 2004, February 12, 2005 and May 23, 2005, respectively. Effluent applications occurred on August 3 and 18, 2004 for Fields 2 and 1, respectively. Data are means across four plots (n = 4) within each field at a sampling date (LSD = 15.5 mg/L).
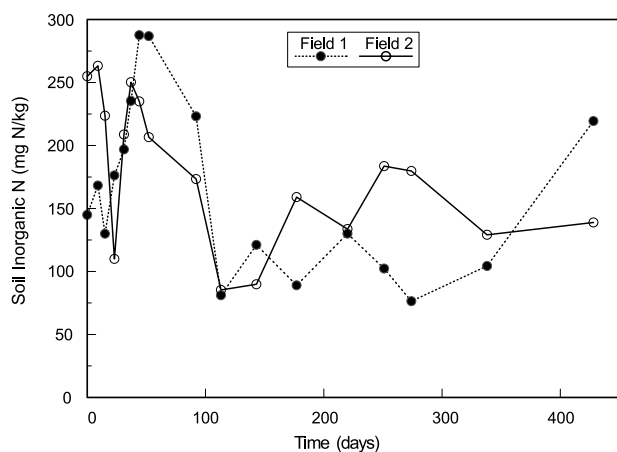

Figure 9. Concentrations of soil inorganic N for samples taken from 0-5 cm depth as a function of time. Time zero is July 27, 2004, date of the first soil sampling. Days 100, 200 and 300 correspond to November 4, 2004, February 12, 2005 and May 23, 2005, respectively. Effluent applications occurred on August 3 and 18, 2004 for Fields 2 and 1, respectively. Data are means across four plots within each field (n = 4) at a sampling date (LSD = 21.7 mg/L).

Most of the soil nitrate and inorganic N was concentrated near the soil surface (Figures 10A and 10B). These data represent the means over the 20 samplings. The average nitrate concentration at the soil surface was exceedingly high in both fields - around 100 mg N/kg. Nitrate levels decreased in the soil profile to a minimum value of around 10 mg N/kg at the 20-40 cm depth, and then increased again as depth increased. Such increases in soil nitrate at depths greater than 40 cm are consistent with the existence of a layer at or below 1 m that was impeding the downward movement of soil water and nitrates. Soil inorganic N (which includes ammonium) followed the same trends (Figure 10B), but was higher, 175 mg N/kg at the surface.
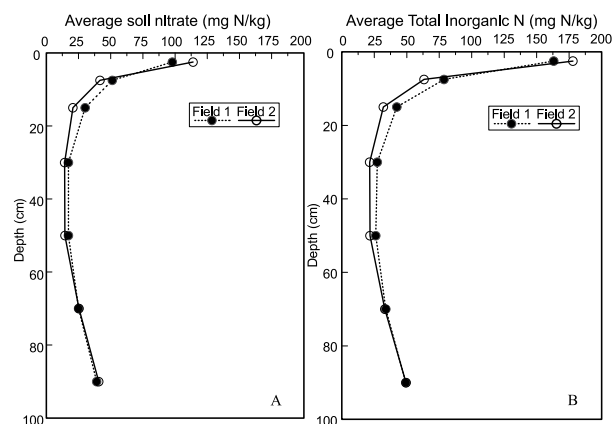

Figure 10. Depth distribution of soil nitrate (panel A) and inorganic N (panel B). Data are means across four plots within each field and 20 sampling dates (n = 80). LSD values for comparing means are 8.34 and 11.6 mg/L for soil nitrate and inorganic N, respectively.

The water soluble soil P values were above 150 mg P/kg (300 pounds/acre) in the surface layer and decreased with depth to near zero at the 60-80 cm depth and below (Figure 11A). These values at the surface are extremely high and represent a "worst-case scenario" for the potential for P losses via run-off. Likewise, Mehlich III values between 1,250 and 1,500 mg P/kg in the surface layer (2,500 to 3,000 lbs P/acre) are higher than necessary for optimum crop growth (Figure 11B). Unlike soil N concentrations, there was no increase in soil P values at lower soil depths. The distribution of soil P in the upper 40 cm was unusual for pastures. Typically, there is a sharp decline in soil P values with depth (Brauer et al., 2005).

In the past, Arkansas has made recommendations for NMPs where liquid manure is applied based on N. Hence, if the soil P values exceeded levels necessary for optimum crop growth, manure applications would still have been allowed. However, the state is

currently changing the policy for farms permitted under Regulation 5 (farms with liquid manure) so that they will have to use the Arkansas Phosphorus Index to determine application rates. At these levels of soil P, manure applications would not be allowed by the Arkansas P Index.
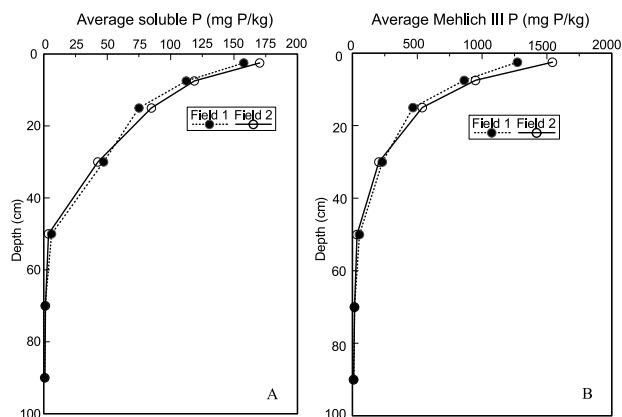

Figure 11. Depth distribution of soluble reactive P (panel A) and Mehlich III P (panel B) concentrations in soil samples. Data are means across four plots within each field and 20 sampling dates (n = 80). LSD values for comparing means are 5.46 and 39.7 mg/L for soluble reactive P and Mehlich III P, respectively.

Forage Data

As in the case of soil samples, one set of forage samples was collected prior to effluent application on Field 2 and three sets of samples were collected from Field 1 prior to effluent application. Nitrate concentrations in forage started out very high (near 3,200 mg NO_3-N/kg forage in Field 1) at the beginning of the study (Figure 12). These high levels of nitrate may have been due to the nitrogen inputs from direct deposit of manures during the winter when the fields were used as a loafing area. Bermudagrass forage is usually considered a low risk for inducing nitrate toxicity in cattle because it does not tend to accumulate enough nitrate to induce toxicity (Strickland et al., 2008). In other studies, nitrate-nitrogen concentrations in bermudagrass forage did not exceed more than 400 mg NO_3-N/kg (dry weight) with N fertilizer application up to 100 kg N/ha (Hojjati et al., 1972). N fertilizer application rates of 200 to 400 kg N/ha increased concentrations to 1,200 to 1,600 kg NO_3-N/kg. Toxicity of cattle to nitrates is low if the forage is less than 3,000 mg NO_3-N/kg (Strickland et al., 2008). Forage nitrate concentrations in samples from Field 1 declined over the first four sampling dates, the last of which were collected the day after effluent application. Forage nitrate concentrations in samples

from Field 1 were higher than the initial levels over the next four sampling dates. These increases in forage nitrate concentrations corresponded to increases in soil nitrate levels in the samples from the surface layer (0-5 cm). An increase in forage nitrate concentrations for samples from Field 2 during the early part of the study was not readily discernible. In general, forage nitrate concentrations decreased to a minimum of 26 mg NO_3-N/kg forage at Day 220 (February 26, 2005) and increased to over 2,000 mg NO_3-N/kg by Day 274 (April 21, 2005) in Field 1. The forage on this site was bermudagrass, which is a warm season grass. In northwest Arkansas, bermudagrass will become dormant around November 1 and will resume growth by May 1. During the winter months, nitrate, as well as other water soluble constituents, is leached from the forage, thus explaining the decrease in nitrate concentrations during the winter.
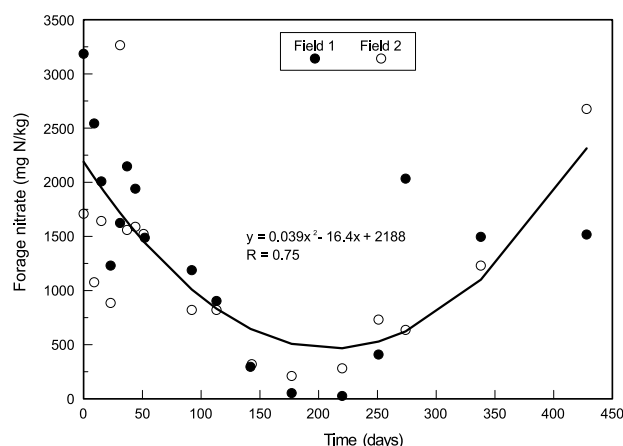

Figure 12. Forage nitrate concentrations as a function of time. Time zero is July 27, 2004, date of the first soil sampling. Days 100, 200 and 300 correspond to November 4, 2004, February 12, 2005 and May 23, 2005, respectively. Effluent applications occurred on August 3 and 18, 2004 for Fields 2 and 1, respectively. Data are means across four plots within a field (n = 4). Equation and regression coefficient (R) appear in figure.

Metrics for Nitrate Leaching

Prior to conducting this study we hypothesized that soil nitrate would be the best metric for predicting nitrate leaching. However, the relationship between soil nitrate concentrations at 80-100 cm and lysimeter nitrate levels was poor (y = -0.79x + 172, r = 0.08 (P > 0.10), data not shown). The best relationship between soil nitrate and lysimeter nitrate was with the samples taken from the 0-5 cm depth (y = 0.56x + 92, r = 0.42 (P < 0.05), data not shown). Although this relationship is statistically significant, its predictive power is poor.

The best indicator of high concentrations of nitrate in soil solution was the forage nitrate concentrations (Figure 13). Many growers often have their hay analyzed for nitrate to make sure there is little or no potential for nitrate toxicity in their cattle.   Forage nitrate concentrations followed the same seasonal patterns as did lysimeter nitrate concentrations, with the highest levels observed at the beginning and ending of study; i.e., during summer (Figure 13).
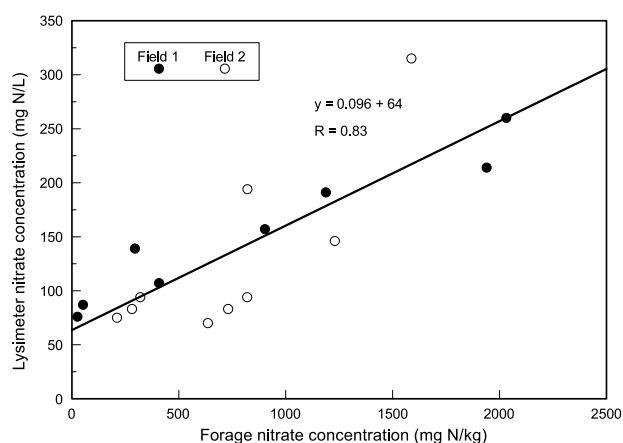

Figure 13. Relationship between lysimeter nitrate and forage nitrate concentrations. Data are means across four plots within a field (n = 4) at eight sampling dates. Regression equation and coefficient (R) are presented in figure.

The lysimeter nitrate concentrations and soil P concentrations observed in this study were orders of magnitude higher than what is normally observed in poultry/beef farms located in northwest Arkansas. The reason for these higher nutrient levels did not appear to be the effluent applications; but rather, they appeared to be due to direct manure deposits from dairy cattle as a result of using the fields as the loafing area for the cows during winter months. Beginning in November the surface of these fields became entirely covered with cow manure. Although the levels of nitrate found in lysimeters were often in excess of 100 mg NO_3-N/L, it is not known if ground-water contamination actually occurred, since deeper wells were not installed to investigate this.

When representatives from USDA/NRCS and University of Arkansas Cooperative Extension Service were interviewed about the mechanics of the NMP for liquid manures using Regulation 5, we discovered that the amount of nutrients deposited directly from the animals was not taken into consideration. Under normal grazing conditions, this practice would not cause a problem, since manure rates from grazing animals is normally less than what is applied via effluent. However, in dairy loafing areas, when dozens of dairy cows consuming a complete ration are stocked at a high number of animals per land area, then excessive nutrient applications can result.

In hindsight, accumulation of N and P in the soils at this farm should have been expected. The 250 dairy cows on this farm would be expected to generate about 21,000 kg N and 2,800 kg P in their manure annually, based on USDA estimates referenced earlier in this report. These amounts of excreted N and P calculate to be annual loading rates in excess of 700 kg N/ha and 100 kg P/ha, assuming that only the 27 ha on the farm were available for deposition.

The use of pastures or hay fields as HUA may be a difficult situation to correct for many dairy farmers. The cows must be kept within certain proximity of the dairy when days are short. In this case, these four fields receiving the effluent were the four closest fields to the milking parlor. The solution would be to spread the cows out over a greater area (make more fields into loafing areas), so that the manure concentrations would be reduced. In addition, the effluent from the holding pond should be applied to other fields that are not used for loafing. This is also problematic, since it requires piping the liquid manure greater distances on the farm, which increases the grower's cost. However, this is one fairly simple solution that is easily implemented. Dr. Karl VanDevender is an Agricultural Engineer with the University of Arkansas Cooperative Extension Service and is in charge of Extension activities involving liquid manure applications in Arkansas. His recommendation for this problem is as follows: "Growers should decide whether they want to keep a field in pasture or use it as a HUA or loafing area. If it is kept in pasture, then the cow density and use duration should be kept down to the point where forage is maintained. These pastures would be subject to Phosphorus Index if additional manure (effluent) is applied, just as they would for a beef cattle operation. If the grower decides to make it a HUA, then he/she should design and manage to 1) minimize the area, 2) divert runoff water while it is clean, 3) scrape/clean surface as needed to avoid low spots, ponding, excessive manure accumulations, and 4) treat filter runoff water from HUA with filter strips and grass waterways." This advice appears to be sound and well grounded. Efforts will also be made to restrict effluent applications on dairy loafing areas in the revised Arkansas P Index. It also appears that annual loading rates of N and P need to be considered at the farm level when NMPs are being developed. If such calculations had been performed in the preparation of this farm's NMP, it should have been obvious that the 31 ha was insufficient land area for application of the amount of waste created to avoid the buildup of nutrients in the soil.

The best metric to be indicative of nitrate leaching was the forage nitrate concentration. Since growers do not want to feed cattle forages containing excessive nitrate levels, they would be willing to pay for forage nitrate analyses. Hence, we recommend that forage nitrate be routinely measured on spray fields receiving effluent to determine if excessive nitrate leaching is occurring. This would also serve a dual purpose of protecting cattle from poor quality vegetation. Forage sampling for nitrate analyses just prior to harvesting would provide indicators of utility of the hay for dairy cows and if future effluent application protocol needed to be adjusted.

5.0
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