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Impact of Large Poultry Operations on Groundwater : Stable ^{15}N Isotopes of Nitrate Assessment

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Abstract: The rapid growth of livestock and poultry operations over the past 25 years has led to concern regarding environmental impacts of land applied animal wastes in many parts of the world. This study investigated the impact of dense poultry populations on ground and surface water quality in central North Carolina using the stable ^{15}N isotopes of nitrate. On a field scale, the ^{15}N of groundwater nitrate was not related to the type of poultry waste applied to the fields, but was controlled by the types of soils found in the litter application fields. Hydric soils had enriched groundwater nitrate ^{15}N compositions and reduced nitrate concentrations, most likely associated with denitrification. Partially hydric and non-hydric soils did not show elevated groundwater ^{15}N nitrate compositions. The ^{15}N composition of groundwater nitrate in litter application fields with non-hydric soils indicates that the source of groundwater nitrate is a combination of poultry litter and fertilizer. On a watershed scale, the ^{15}N composition of surface water nitrate was not related to the distribution of poultry operations, but was related to the distribution of hydric soils within the basin. Nitrate concentrations in stream waters remained low through out the basin studied, and the ^{15}N composition of stream nitrate decreased downstream suggesting minimal impact on surface water quality from the surrounding poultry operations. The practice of placing poultry houses on poor quality swampy land also places them in areas dominated by hydric soils. Denitrification proceeds quickly in areas with hydric soils, which minimizes offsite nitrate transport and mitigates surface water quality impacts. This data suggests that surface water quality impacts from animal agriculture can be predicted from the spatial analysis of hydric soils within a watershed.

Key words: ^{15}N natural abundance, poultry litter, denitrification, watershed water quality

Introduction

Human activity has altered the nitrogen cycle more than any other element (Vitousek *et al*., 1997, Howarth, 2004). Nitrogen fixation by human activity (industrial and agricultural) is currently estimated at over 160 Tg N per year, which is 30 to 45% of the total nitrogen fixed on land and in the oceans (Vitousek *et al*., 1997, Cleveland *et al*., 1999, Karl *et al*., 2002, Howarth 2004). On land, anthropogenically mobilized reactive nitrogen (N_r) now equals the amount of nitrogen naturally fixed by terrestrial ecosystems (Galloway *et al*., 2003). Nitrogen pollution from this additional N_r results in acidification of soils, loss of biodiversity in terrestrial and aquatic ecosystems, and eutrophication of coastal waters (Howarth, 2004). Agriculture is the largest driver of anthropogenic changes in the global nitrogen cycle. Inorganic fertilizer applied to agricultural lands accounts for 90 Tg N per year of the total N fixed by human activity, and another 30 Tg N per year is fixed in agro-ecosystems (Howarth, 2004, Galloway *et al*., 2003). 60 Tg N per year is harvested as crops, and the rest of the N_r leaches into surface and ground waters (Smil, 2002, Howarth *et al*., 2002a, 2002b, van Breeman *et al*., 2002). Humans consume 27% of the crops harvested, and approximately 55% of the harvested crops are fed to animals (~33 Tg N per year). Of this 33 Tg N per year globally consumed by animals, ~5 Tg N per year is subsequently converted to human food (Smil, 2002). The remaining 28 Tg N per year is released back to the environment as animal wastes (manure and urine), or as wastes lost during slaughtering and processing (Smil, 2002; Howarth, 2004). Therefore, 25% of the global N used in agriculture becomes the waste by-product of animal production systems. This animal by-product waste can leach into ground and surface waters, or can be volatilized back into the atmosphere as ammonia (Holland *et al*., 1999; Howarth *et al*., 2002a). Tracing the environmental impacts of animal production systems are problematic, but an environmentally and economically important issue facing modern agriculture today.

In North Carolina there has been rapid growth in food-animal populations over the past 25 years (Fig. 1). Poultry populations expanded in the 1980's, while then

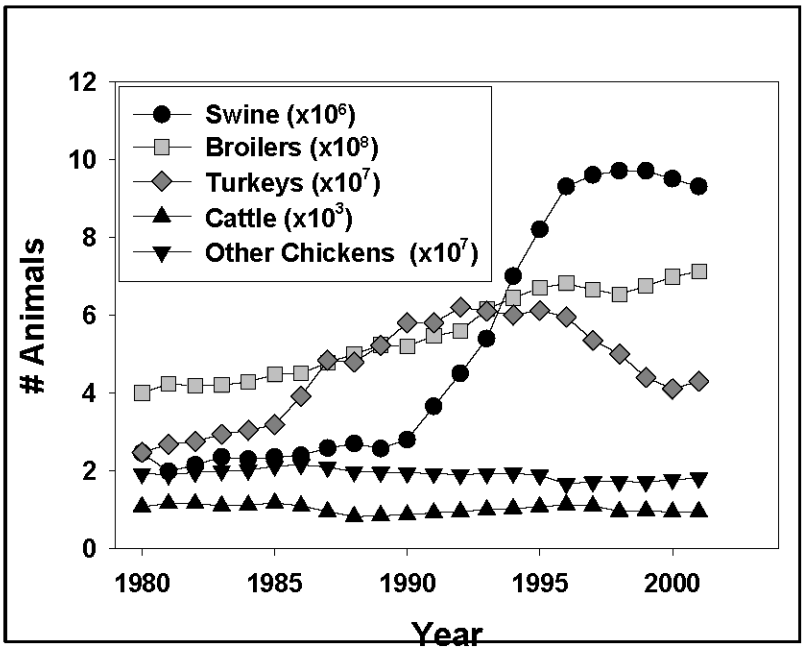

Fig. 1: NC animal populations over the past 25 years. Poultry populations grew in the 1980's, while the swine population grew in the 1990's.

swine populations grew dramatically in the 1990's. The issue of environmental impacts of CAFO's and animal waste disposal in North Carolina has become one of the top environmental concerns, and an impediment to the continued growth of the food-animal production industry. It is well established that agricultural activities have large effects on groundwater recharge rates and chemical loads (Böhlke, 2002). Agricultural contaminate loads in recharged groundwater can pose health risks when nitrate concentrations are above 10 mg/l in drinking water (Fan and Steinberg, 1996, Cantor, 1997, Böhlke, 2002, Gulis *et al.*, 2002). Complicated groundwater flow paths that are site specific can result in different sources of N accumulating in groundwater as land use changes. Watershed NO_3 gradients that depend locally on transient and steady state hydro-geological variables are not well documented and hard to interpret (Böhlke and Denver, 1995). Large N fluxes to shallow groundwater systems in agricultural areas are also commonly not matched by N discharge rates in adjacent streams (Böhlke and Denver, 1995, Böhlke, 2002). On a watershed scale, stream flow N exports generally account for only ~25% of total N inputs to the basin (Howarth *et al.*, 1996, Boyer *et al.*, 2002, van Breemen *et al.*, 2002). The "missing" nitrogen can be retained in landscapes as wood, soil organic N or stored temporarily in ground waters with residence times of decades to centuries (van Breeman *et al.*, 2002). Watershed N can also be lost as N_2 to the atmosphere during denitrification. Attempts to estimate the amount of N loss from denitrification by the estimating the difference between all inputs, storage and loss terms suggest that up to 50 % of N inputs are consumed by denitrification (Boyer *et al.*, 2002, van Breeman *et al.*, 2002). Climatic factors such as droughts and floods can also affect riverine N export, and these inter-annual changes can affect N watershed fluxes more than discharge variations (Justic *et al.*, 2003). N loss due to denitrification from watersheds on the field to watershed scales is important, but not well documented because of spatial and temporal variability.

Identification of nitrogen sources that contribute to water quality degradation is crucial for management response and regulation of water quality issues. Nitrogen isotope techniques have been used to identify nitrogen sources and describe N transformations (denitrification) in terrestrial and aquatic ecosystems (Letolle, 1980, Hubner, 1986; Kendall, 1998; Kendall and Aravena, 2000). Nitrate derived from manure or sewage is enriched in the heavier ^{15}N isotope due to ammonia volatilization, and has $\delta^{15}N$ values between +7 to > +20% (Kreitler and Jones, 1975; Gormly and Spalding, 1979, Kreitler, 1979, Aravena *et al.*, 1993, Wassenaar, 1995, Aravena and Robertson, 1998, Karr *et al.*, 2001, Showers *et al.*, 2005). Waste N is distinct from N derived from atmospheric deposition (-10 to +5%), N in fertilizers (0±3%), and from natural soil organic N (-3 to +5%; Mayer *et al.*, 2002, Kendall, 1998, Kendall and Aravena, 2000). Kellman and Hillaire-Marcel (1998) reported significant increase in $\delta^{15}N$ NO_3 in small agricultural streams, which they attributed to denitrification. Mayer *et al.* (2002) found that the isotopic composition of nitrate

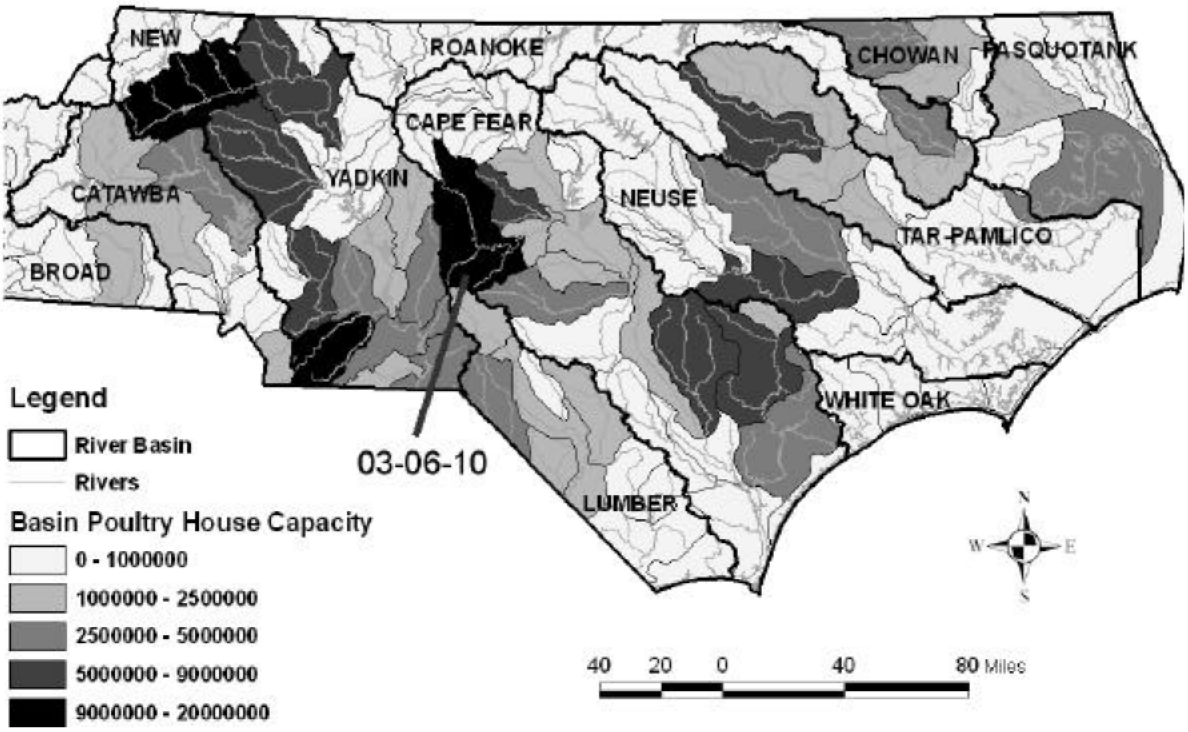

Fig. 2: Poultry house capacity by river sub-basin for North Carolina (1996). Surface water stations in sub-basin 03-06-10 from the Cape Fear River Basin was sampled over a 1 year period to determine poultry operation impacts on surface water quality (see Fig. 5a).

reflected the contribution of wastewater and manure to the nitrogen load in 16 large watersheds in the northeastern US, but did not detect significant amounts of denitrification in these surface river waters. Showers *et al.*, (2005) found that dual isotopic indicators (^{15}N and 18O) indicated denitrification in groundwater under biosolid application fields in hydric soils while non-hydric soils did not have elevated isotopic compositions. This study also suggested that the presence or absence of hydric soils in waste application fields control the amount of nitrogen exported from the field to adjacent surface waters. This paper examines the distribution of hydric soils and the nitrate isotopic composition of groundwater in poultry litter application fields. On a larger scale, the nitrate isotopic composition of surface waters in a watershed with a large poultry population was also analyzed to determine the impacts on surface water quality of dense poultry operations in central North Carolina.

Materials and Methods

Groundwater samples were collected from six different poultry litter application fields at four different farms during a one-year period in Bladen and Hoke Counties, NC. Two to four wells were drilled 8 to 40 feet down to the depth of the water table in poultry litter application fields. Wells were purged of 3-5 volumes of water, then 1 liter samples were collected in acid washed (0.1 M HCl) Nalgene bottles. Surface water samples also were collected in acid washed 1 liter Nalgene bottles from adjacent creek or streams, and from 14 surface water stations in the Moore County Richlands Creek watershed (sub-basin 03-06-10) which had a poultry population (house capacity) of 9.4 million birds in 1996 (Fig. 2). This is one of the densest concentrations of poultry in any NC watershed. Water samples were filtered with a GWV 63 micron filter and kept at 4°C until processed for nutrient concentrations and isotopic abundances.

In the lab, surface and groundwater samples were filtered through a Gelman AquaPrep 600 cartridge filter (0.45 micron) or a GFF precombusted filter (0.77 micron, heated to 500°C for 4 hours). Nutrient concentrations (NO_3, NH_4, PO_4) were determined on the filtered samples. Approximately 10 ml of the filtered water was analyzed in an automated flow injection La Chat Quick-Chem 8000 Ion Chromatograph (IC) for nitrate+nitrite (EPA Method 353.2, USEPA, 1993), phosphate (EPA Method 365.1, USEPA, 1993) and ammonium (EPA Method 350.1, USEPA, 1993). During each La Chat IC run, an external standard (EPA) and several internal QC standards were run with 10 dilution standards and one spiked water sample to quantify matrix effects. An additional internal QC standard was run for every 10

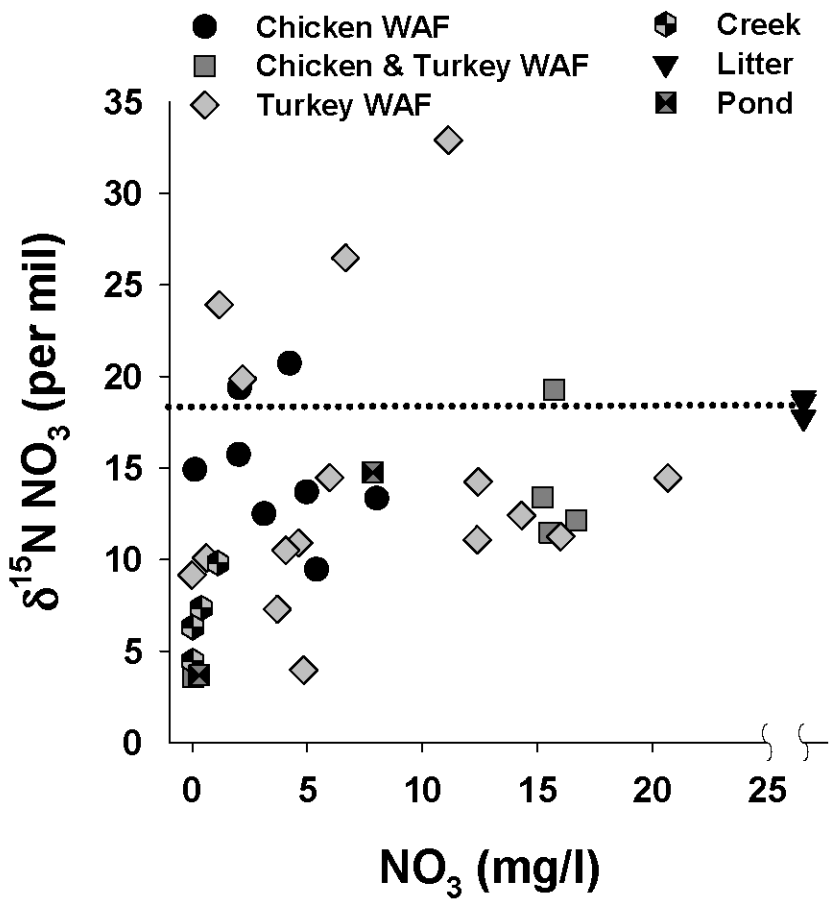

Fig. 3: Nitrate concentration and isotopic composition of surface and ground water samples taken at poultry farms in Hoke and Bladen Counties, North Carolina. Total N isotopic composition of poultry litter is plotted on the right without N concentration.

samples analyzed. The $\delta^{15}N$ of dissolved nitrate or ammonia was analyzed by a modification of the technique of Chang *et al.* (1999). 1-4 liters of sample, which was enough water to yield 15 µM of nitrogen, were passed through a double ion exchange resin column (1st - cation - 5 ml Biorad AG 50-WX8; 2nd - anion - 2ml Biorad AG 2-X8). The cation column was pre-washed with deionized water. The anion column was pre-washed with 3N HCL, and then repeatedly washed with deionized water to remove all acid residues. Pre-washing the anion column with the same strength acid as the elutant allows 15 µM dissolved N samples to be analyzed without an isotopic correction (Showers *et al.*, 2005). Nitrate was eluted from the anion column with 30 ml of 3N HCl. The HCl was neutralized with 15 gm of Ag_2O, the sample was filtered with a Whatman GFF filter, and the filtrate was freeze dried to yield a fine white powder of $AgNO_3$. Half the sample was placed in a tin boat and combusted in a Carlo Erba NC2500 Elemental Analyzer and isotopically analyzed with a Finnigan Mat Delta+ XL CF-IRMS to determine $\delta^{15}N$-NO_3. The $\delta^{15}N$ results were calibrated against NIST 8550, NIST 8548, NIST 8547, and four internal ^{15}N standards. Statistical analysis of all the river flux, nutrient concentration, and isotopic results were completed with Microsoft Excel spreadsheets. GIS analysis of soil types was done with ArcInfo 9.1 and County Soil GIS data sets from NC CGIA (North Carolina Center for Geographic Information and Analysis) at a 1:24,000 scale (USDA NRCS, 1998). In the late 1990s, the North Carolina Department of Environment and Natural Resources, Division of Water Quality developed a GIS data set representing point locations of CAFO's in North Carolina. In 2003, CGIA used digital ortho-photography from 1998 to verify the DWQ locations for swine operations and identified all swine waste lagoons visible in the photos. For the McLendon's Creek and Richlands Creek sub-basins, poultry barns were also identified from the 1998 infra-red 1:12,000 digital orthophoto quarter quadrangle (DOQQ data set). Each poultry house data point represents 1 to >10 barns without a lagoon that are not identified in the 2003 CGIA data set as a swine operation.

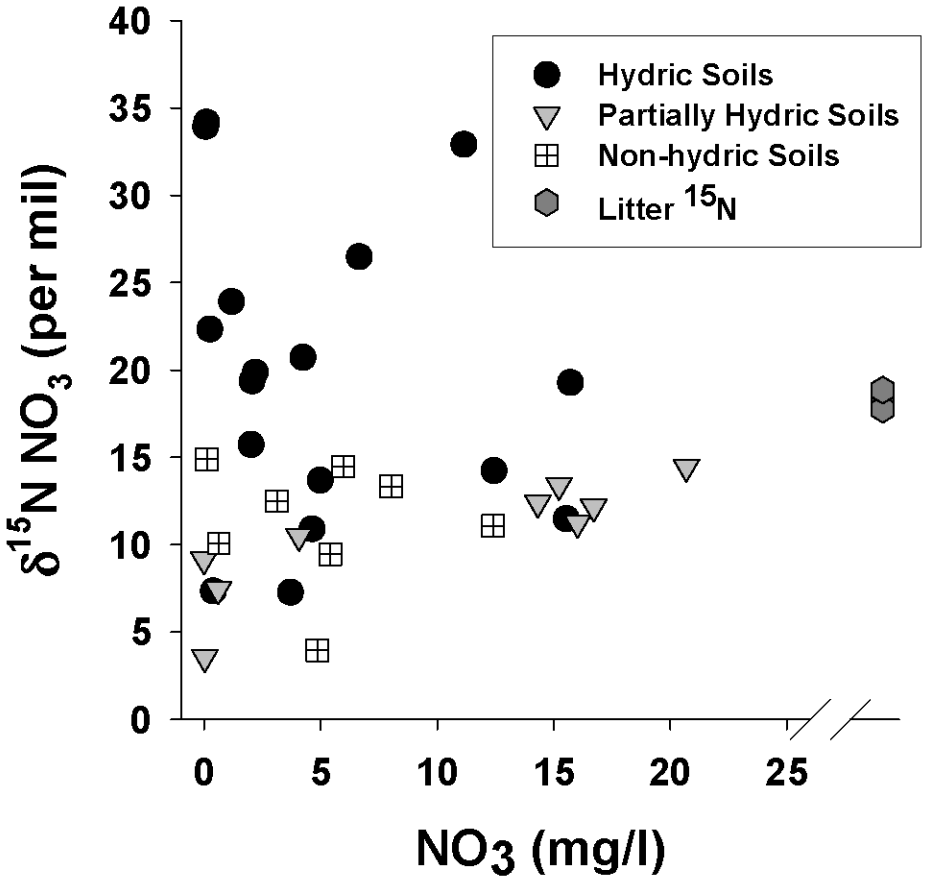

Fig. 4: Soil type ^{15}N composition plotted against nitrate concentration for ground waters in poultry litter application fields.

Results

The nitrate concentration in ground waters under poultry litter application fields varied from 0 to 21 mg/l (Fig. 3). The nitrate $\delta^{15}N$ in ground waters under poultry litter application fields varied from +3 to +34 per mil. Chicken and turkey litter samples collected from the house floor varied between +17 to +19 per mil $\delta^{15}N$. Creek samples collected in surface drainages adjacent to the litter application fields varied from +4 to +9 per mil $\delta^{15}N$, and had nitrate concentrations less than 1.2 mg/l (Fig. 3). The two pond surface water samples were very different. One pond had a nitrate concentration of 8 mg/l, and a $\delta^{15}N$ nitrate composition of +15 per mil. The second pond had a nitrate concentration of less than 1 mg/l, and a $\delta^{15}N$ nitrate composition of 4 per mil (Fig. 3). There appeared to be no concentration or isotopic differences between fields that received different types of poultry litter (chicken, turkey, or chicken plus turkey; Fig. 3).

Approximately 30% of the wells sampled have ^{15}N compositions above the nitrogen isotopic composition of the poultry litter that was applied to the fields. Spatial analysis of the soil type at the location of the monitoring wells revealed a complicated distribution of hydric soils in the application fields. Within one field hydric, partially hydric, and non-hydric soils could be present. The groundwater nitrate ^{15}N compositions are correlated to the soil hydric classification. All of the wells with elevated ^{15}N compositions were located in hydric or partially hydric soils (Fig. 6). All of the wells located in non-hydric soils had ^{15}N nitrate isotopic compositions below the ^{15}N composition of poultry litter applied to the fields. At a Hoke County Breeder site two ponds were located in litter application fields. One pond located in hydric soils had an elevated ^{15}N nitrate composition. The other pond located in non-hydric soils has a low ^{15}N nitrate isotopic composition. The distribution of hydric soils correlates to groundwater nitrate ^{15}N at all the groundwater sampling sites.

Surface waters in the McLendon's Creek and Richland Creek watersheds were sampled over a 12-month period from February 1998 to January 1999 at ten separate stations (Fig. 5a). Six stations were located in the lower portion of the basin with abundant hydric and partially hydric soils. One lower basin station was located on a small creek next to a swine operation (Fig. 5a). This sub-basin is ~101 square miles in area. In

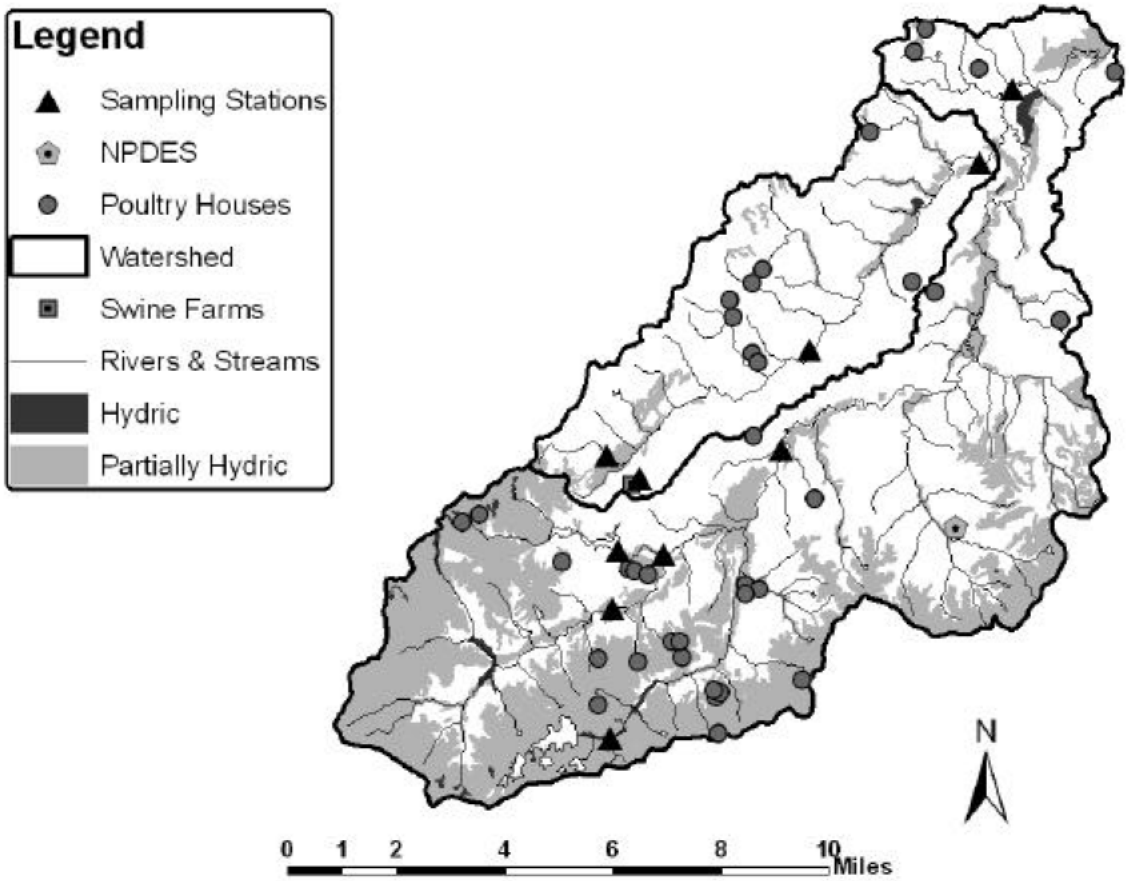

Fig. 5a: McLendon's Creek watershed with river sampling stations, poultry houses and the distribution of hydric and partially hydric soils.

2002, land use in the sub-basin was predominately forested (79%), shrubland (12%) and wetlands (5%). Cultivated and urban land usage was less than 2% each. Poultry houses are evenly distributed across the sub-basin (Fig. 5). Discharge measured at a USGS gauging station (#2102000) immediately downstream from the sub-basin was high in the winter and in the low summer (Fig. 5b). The USGS discharge record indicates that discharge during the sampling period was not markedly different from previous years, and suggests that the samples collected during this study were representative of normal hydrological conditions in the basin. Spatial analysis of soil types in the basin showed a marked difference between the southern (upstream) and northern (downstream) areas of the basin. Hydric and partially hydric soils are abundant in the lower portion of the watershed, while the upper portion of the watershed has mostly non-hydric soils (Fig. 5a). Surface water nitrate concentrations in these creeks varied from 0 to 1 mg/l, except at station 15 adjacent to the swine CAFO, which had higher nitrate concentrations (0.2 to 4 mg/l; Fig. 6). ^{15}N nitrate surface water compositions were distinctly different between stations located in hydric soils in the lower basin and non-hydric soils in the upper basin (Fig. 6). The upper sub-basin stations (MC 1-8) located in hydric and partially hydric soils had ^{15}N surface water nitrate compositions of +4 to +20% . The lower basin stations (9-11) located in non-hydric soils had ^{15}N compositions from +4 to+10%. The stream adjacent to the swine CAFO (MC 15) had surface water nitrate ^{15}N compositions that varied from +11 to +28%. When the average surface water nitrate concentration and ^{15}N compositions are plotted on a basin scale, general basin-wide water quality trends can be assessed. The average surface water nitrate concentration does not significantly change downstream, but the average ^{15}N composition decreases from +10 per mil to +6% in the lower basin (Fig. 7).

Discussion

Poultry litter collected from turkey and chicken houses have a uniform ^{15}N composition ranging from +17 to +19 per mil. The elevated ^{15}N composition is the result of ammonia volatilization as the litter dries (Kendall, 1998). By comparison swine wastes that are contained in liquid

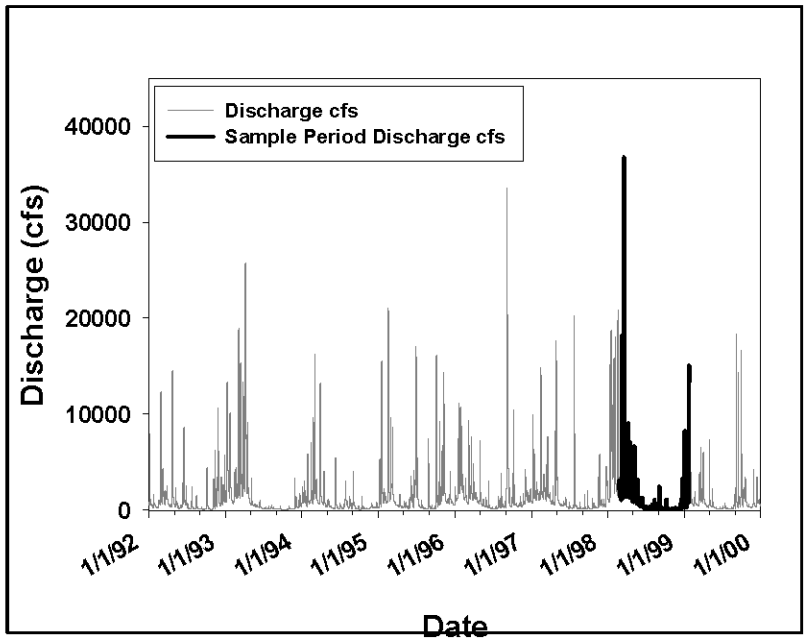

Fig. 5b: USGS discharge record at Station 2102000 on the Deep River at Monclure, NC directly downstream from Mclendon's Creek sub-basin. The period sampled in marked by the dark lines.

lagoons have large seasonal ^{15}N variations, because volatilization of ammonia that enriches ^{15}N in the liquid waste is temperature dependent (Karr *et al.*, 2001). Poultry litter quickly dries on the house floor, and once dry further ^{15}N enrichment of the litter is not likely to continue resulting in a fairly uniform ^{15}N litter composition over time. Groundwater nitrate under the litter application fields had a wide variation of ^{15}N compositions that do not correlate to the type of poultry litter applied (chicken, turkey, chicken plus turkey). Spatial analysis of the hydric soil type indicate that there is a wide variety of soil types present in one application field in central North Carolina. Hydric, partially hydric and non-hydric soils may all be located in one field. Elevated groundwater nitrate ^{15}N values, however, are only found in litter application fields monitoring wells with hydric soils. Denitrification is likely responsible for the elevated ^{15}N nitrate values found in hydric soils, because hydric soils are wet, anoxic, and carbon rich areas. Wells in non-hydric or partially hydric soils had groundwater nitrate ^{15}N values that are not elevated, suggesting that the source of nitrate in these wells is from fertilizer mixing with poultry litter nitrogen in various proportions to produce intermediate ^{15}N compositions (+3 to +15%). Nitrate concentrations in poultry litter fields sampled in this study varied from 0.1 to 23 mg/l. These concentrations are significantly lower than nitrate concentrations measured in North Carolina swine, dairy, and biosolid waste application fields, which can have nitrate concentrations over 100 mg/l (Karr *et al.*, 2001, 2002; Showers *et al.*, 2005). Poultry application fields sampled in this study are located in areas with abundant hydric soils. The correlation of elevated nitrate ^{15}N and low nitrate groundwater concentrations associated with hydric soils suggests that natural attenuation by denitrification has mitigated environmental impacts from poultry litter applications in central North Carolina on a field scale.

On a basin scale, nitrate concentrations in the McLendon's and Richland Creek watersheds were low except in a small stream adjacent to a swine CAFO. Poultry houses in this watershed are uniformly spread throughout the basin. The nitrate concentrations do not change significantly downstream in this basin, but the ^{15}N composition of nitrate actually decreases downstream. The surface waters sampling stations in hydric soils have consistently heavier ^{15}N compositions of nitrate than stations in the lower basin located in non-hydric soils. This basin-wide nitrate concentration and ^{15}N surface water pattern is inconsistent with the isotopically heavy poultry litter nitrogen accumulating in the drainages downstream. Accumulation of poultry litter N downstream would result in the higher concentrations and heavier ^{15}N nitrate in the lower basin. This data indicates that the distribution of hydric soils, and not density and location of poultry operations controls water quality trends in this basin, even though over 9.3 million birds are located in an area of ~101 square miles.

On a watershed scale heavy ^{15}N in surface water nitrate has been interpreted to indicate the input of waste

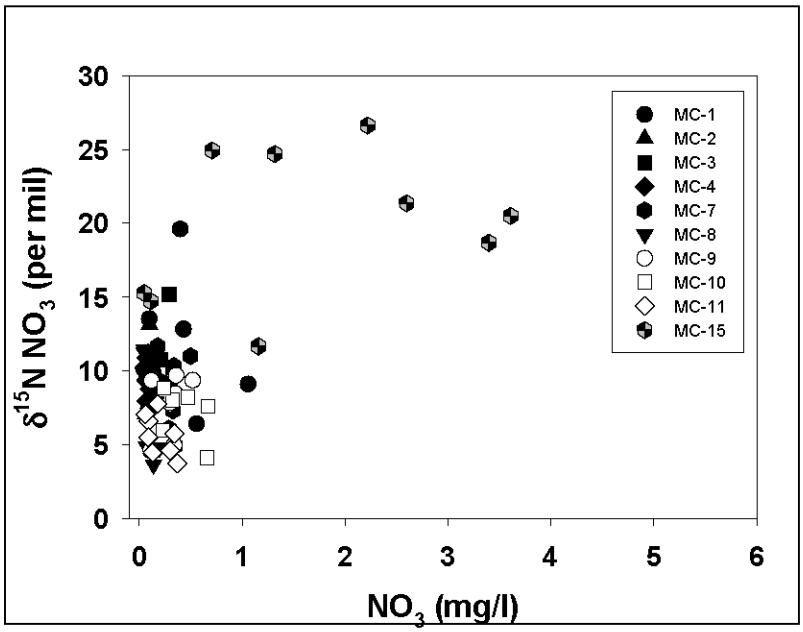

Fig. 6: Nitrate concentration and isotopic composition of surface sampling stations in the McLendon's Creek watershed. Filled symbols are stations located in hydric soils, open symbols are stations located in non-hydric soils. The hexagonal cross hatched symbols is Station #15 which is located in non-hydric soils adjacent to a swine CAFO.

nitrogen into watersheds (McKinney *et al.*, 2002; Mayer *et al.*, 2002). On smaller field scales, heavy ^{15}N in surface water nitrate draining fertilized agricultural fields has been interpreted to be the result of extensive denitrification in tile drain systems (Kellman and Hillaire-Marcel, 1998). Increasing nitrate contamination of surface and groundwater is a common problem in regions of intensive agriculture (Spalding and Exner, 1993), but linking field application rates to surface water quality trends is problematic because of complicated groundwater flow paths (Böhlke and Denver, 1995, Böhlke, 2002). On a watershed scale, basin N discharge rates are usually significantly lower than the sum of all N inputs to the basin (Howarth, 2002b, van Breemen *et al.*, 2002). Riverine nitrogen fluxes typically only accounts for ~25% of the nitrogen input into watersheds (Kendall, 1998; Cane and Clark, 1999; Kendall and Aravena, 2000). Estimates of denitrification calculated from the difference between all the N inputs and N exports in many basins suggest that denitrification may account for over 50% of N loss on watershed scales (van Breemen *et al.*, 2002). But spatial variation of denitrification on basin scales is not understood. These ^{15}N nitrate results indicates that the spatial distribution of hydric soils is controlling water quality in surface waters and ground waters on field to basin-wide scales.

Land use in the McLendon's Creek sub-basin is dominated by forests and poultry operations. Since forests do not export large amounts of nitrogen, poultry operations should be the dominant source of nitrate in the streams in this basin. Accumulation of poultry waste nitrogen in surface drainages is not consistent with the pattern of low nitrate concentrations and decreasing ^{15}N nitrate downstream. Several factors can control ^{15}N in ground and surface water nitrate including ^{15}N variations of the sources (inputs) and sinks (outputs), mixing of point and non-point sources along flow paths for groundwater, as well as transformation of nitrogen (denitrification) within the soils or groundwater (Howarth 2004). If poultry litter applied to fields in the watershed were transferring animal waste nitrate into the adjacent streams, the isotopic composition of streams in the upper basin should be ~15 $^{o}/_{oo}$, and both nitrate concentrations and ^{15}N should increase downstream. High nitrate ^{15}N in the upper basin and low concentrations suggest that the presence of hydric soils and denitrification is controlling the stream ^{15}N nitrate compositions. Poultry litter nitrogen cannot be the dominant source of nitrate to the creeks, because the observed stream ^{15}N nitrate values are below the isotopic composition of poultry litter collected in the basin. Surface water nitrate concentrations stay low in the lower basin, but the ^{15}N nitrate values decrease downstream. This is consistent with N exported from forests (Kendall, 1998), but raises the question why such a dense concentration of poultry does not have an observable impacts on surface water quality.

15N and ^{18}O of nitrate has been used to identify the

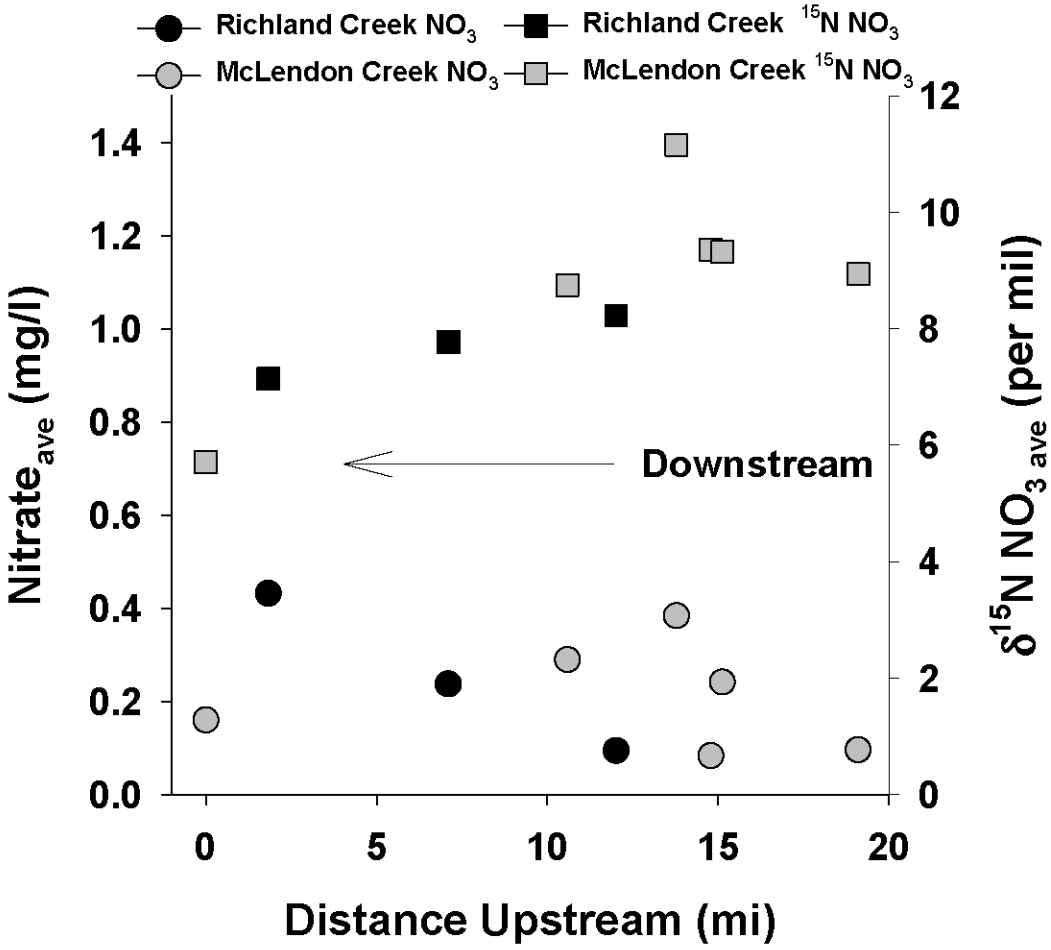

Fig. 7: Isotopic composition and nitrate concentration of Richland and McLendon's Creeks plotted as distance upstream from the lowest station in the basin.

location of denitrification in ground waters (Aravena *et al.*, 1993; Panno *et al.*, 2001; Fukada *et al.*, 2003). Showers *et al.* (2005) used this relationship in biosolid application fields at a large municipal wastewater treatment plant to determine that areas of denitrification in application fields were associated with hydric and partially hydric soils. Non-hydric soils did not have elevated groundwater ^{15}N and ^{18}O values. Israel *et al.*, (2005) determined that denitrification was not important in riparian buffer areas adjacent to a large swine operation in the Six Runs Creek watershed using the dual ^{15}N and ^{18}O tracer approach. They found that waste nitrate with ^{15}N compositions near average waste lagoon values migrated into adjacent surface waters after 5 years of operation in non-hydric buffers. Poultry litter application fields examined in this study are located in areas of abundant hydric soils. Elevated ^{15}N nitrate compositions were found in ground waters located in hydric soils. Kellman and Hillaire-Marcel 1998 found elevated ^{15}N nitrate in small streams draining a heavily fertilized watershed in the St Lawrence lowlands near Quebec Canada. They suggest that a shift of 10% of ^{15}N nitrate in these streams correspond to a 50% nitrate loss of nitrogen applied to the fields. Ground waters under the poultry litter application fields in hydric soils examined in this study have up to a 15 per mil ^{15}N shift, assuming that all the nitrate in the groundwater under the application fields comes from poultry litter. In non-hydric areas, litter application field groundwater nitrate 15 gm of Ag_2O% indicating a mixture of fertilizer and poultry litter nitrogen in groundwater. Using a Rayleigh equation to calculate N loss and source end members of 5 to 20 mg/l and ^{15}N values of +7% to +17% , ^{15}N nitrate values of +20% to +35% observed in hydric soils represents an N loss of 60 to 95%. The amount of N loss calculated from Rayleigh equations is well correlated to the enrichment of ^{15}N in groundwater nitrate and relatively insensitive to the original nitrate concentration. The final ^{15}N composition after near complete denitrification is somewhat sensitive to the original ^{15}N composition. Given the isotopic variation of the sources observed in non-hydric soils, the amount of

N loss from these calculations can vary ± 5% at any particular ^{15}N enrichment. This data suggests significantly greater amounts of denitrification occurs in the hydric soils of North Carolina than the 50% N loss calculated for the St Lawrence lowlands (Kellman and Hillaire-Marcel, 1998). This is consistent with the climatic differences between the two sites, because denitrification rates are related to average temperature (Kendall, 1998). The higher rates of denitrification in central North Carolina is also consistent with the downstream ^{15}N and nitrate concentrations patterns in surface water.

Stream waters in this watershed had low nitrate concentrations and low ^{15}N nitrate compositions suggesting waste nitrogen from the poultry operations is not exported into surface waters despite the dense poultry populations and relatively small basin size. Locating poultry houses in the poor soils with swampy surrounding areas, which contain abundant hydric soils, appears to have mitigated litter nitrate export into adjacent streams. In the absence of other significant nitrogen sources in the basin, the primary inputs into the surface drainages are from the forested areas with a small export from the poultry operations in non-hydric areas. By doing a spatial analysis of hydric soils in watersheds that contain animal operations, environmental impacts of animal waste applications may be able to be predicted.

Conclusions: The ^{15}N composition of poultry litter collected in central North Carolina were consistent and only varied from +17 to +19 per mil. Soil types in poultry litter application fields in this area are heterogeneous on relatively small scales less than the size of the application fields. The hydric class of the soil correlates to the ^{15}N composition of nitrate in groundwater in these litter application fields, and can change over short distances. The significant amount of ^{15}N enrichment observed in groundwater nitrate in hydric soils compared to non-hydric soils indicates that denitrification occurs in hydric soils in this watershed. In hydric soils, the ^{15}N nitrate compositions are high and are controlled by denitrification, so the source of nitrate in these areas cannot be determined. In non-hydric soils where denitrification has not altered the ^{15}N values, the ^{15}N composition of groundwater nitrate indicates a mixture of fertilizer and poultry litter nitrogen in groundwater. Nitrate concentrations under the litter application fields are low compared to other types of animal operations in North Carolina. In the McLendon's Creek watershed, nitrate concentrations and the ^{15}N composition of surface water nitrate decreases downstream. This suggests that the dense poultry populations in the basin do not have a significant impact on surface water quality. The abundant hydric soils in the basin coupled with the lack of other nitrate sources may explain these observations. This study adds to a growing set of observations that the spatial variability of hydric soils control surface and groundwater quality in watersheds with different types of land applied wastes.

Referneces

Aravena, R., M.L. Evans and J.A. Cherry, 1993. Stable isotopes of oxygen and nitrogen in source identification of nitrate from septic systems. Ground Water, 31: 180-186.

Aravena, R. and W.D. Robertson, 1998. Use of multiple isotope tracers to evaluate denitrification in groundwater: case study of nitrate from a large-flux septic system plume, Ground Water, 36: 975-982.

Böhlke, J.K. and J.M. Denver, 1995. Combined use of groundwater dating, chemical, and isotopic analyses to resolve the history and fate of nitrate contamination in two agricultural watersheds, Atlantic Coastal Plain, Maryland. Water Res., 9: 2319-2339.

Böhlke, J.K., 2002 Groundwater recharge and agricultural contamination. Hydrogeol. J., 10: 438-439.

Boyer, E.W., C.L. Goodale, N.A. Jaworsk, R.W. Howarth, 2002. Anthropogenic nitrogen sources and relationships to riverine nitrogen export in the northeastern USA. Biogeochem., 57: 137-169.

Cane, G. and I.D. Clark, 1999 Tracing ground water recharge in an agricultural watershed with isotopes. Ground Water, 37: 133-139.

Cantor, K.P., 1997 Drinking water and cancer. Cancer Causes & Control, 8: 292-308.

Chang, C.C.Y., J. Langston, M. Riggs, D.H. Campbell, S.R. Silva and C. Kendall, 1999. A method for nitrate collection for delta N-15 and delta O-18 analysis from waters with low nitrate concentrations Can. J. Fish. and Aquat. Sci., 56: 1856-1864.

Cleveland, C.C., A.R. Townsend, D.S. Schimel, H. Fisher, R.W. Howarth, L.O. Hedin, S.S. Perakis, E.F. Latty, J.C. Von Fischer, A. Elseroad and M.F. Wasson, 1999. Global patterns of terrestrial biological nitrogen (N-2) fixation in natural ecosystems. Global Biogeochemical Cycles, 13: 623-645.

Fan, A.M. and V.E. Steinberg, 1996 Health implications of nitrate and nitrite in drinking water: An update on methemoglobinemia occurrence and reproductive and developmental toxicity. Regulatory Toxicology and Pharmacology, 23: 35-43.

Fukada, T., K.M. Hiscock, P.F. Dennis and T. Grischek , 2003. A dual isotope approach to identify denitrification in groundwater at a river-bank infiltration site. Water Res., 37: 3070-3078.

Galloway, J.N., J.D. Aber, J.W. Erisman, S.P. Seitzinger, R.W. Howarth, E.B. Cowling and B.J. Cosby, 2003. The Nitrogen Cascade. Biosci., 53: 341-356.

Gormly, J.R. and R.F. Spalding, 1979. Sources and concentrations of nitrate-nitrogen in ground water of the Central Platte Region, Nebraska. Ground Water, 3: 291-301.

Gulis, G., M. Czompolyova and J.R. Cerhan, 2002. An ecologic study of nitrate in municipal drinking water and cancer incidence in Trnava District, Slovakia. Environ. Res., 88: 182-187.

Holland, E.A., F.J. Dentener, B.H. Braswell and J.M. Sulzman, 1999. Contemporary and pre-industrial global reactive nitrogen budgets. Biogeochem., 46: 7-43.

Howarth, R.W., 2004. Human acceleration of the nitrogen cycle: drivers, consequences, and steps toward solutions. Water Sci. Tec., 49: 7-13.

Howarth, R.W., A. Sharpley and D. Walker, 2002a Sources of nutrient pollution to coastal waters in the United States: Implications for achieving coastal water quality goals. Estuaries, 25: 656-676.

Howarth, R.W., E.W. Boyer, W.J. Pabich and J.N. Galloway, 2002b. Nitrogen use in the United States from 1961-2000 and potential future trends. AMBIO, 31: 88-96.

Howarth, R.W., G. Billen, D. Swaney, A. Townsend, N. Jawarski, K. Lajtha, H.A. Downing, R. Eimgren, N. Caraco, T. Jordan, F. Berendse, J. Freney, V. Kudeyaroww, P. Murchoch and Z. Shao-Lian, 1996. Regional nitrogen budgets and riverine N&P fluxes for the drainages to the North Atlantic Ocean:natural and human influence. Biogeochem., 35: 7755-139.

Hubner, H., 1986 Isotope effects of nitrogen in the soil and biosphere. In: Fritz, P. and Fontes, J.Ch. (Eds) Handbook of Environmental Isotope Geochemistry, v2, Elsevier, New York, pp. 361-425.

Israel, D.W., W.J. Showers, M. Fountain and J. Fountain, 2005. Nitrate movement in shallow ground water from swine-lagoon-effluent spray fields managed under current application regulations. J. Environ. Qual., 34: 1828-1842

Justic, D., N.N. Rabalais and R.E. Turner, 2003. Simulated responses of the Gulf of Mexico hypoxia to variations in climate and anthropogenic nutrient loading. J. Marine Sys., 42: 115-126.

Karl, D., A. Michaels, B. Bergman, D. Capone, E. Carpenter, R. Letelier, F. Lipschultz, H. Paerl, D. Sigman and L. Stal, 2002. Dinitrogen fixation in the world's oceans. Biogeochem., 57: 47.

Karr, J.D., W.J. Showers, J.W. Gilliam and A.S. Anders, 2001 Tracing nitrate transport and environmental impact from intensive swine farming using delta nitrogen-15. J. Environ. Qual., 30: 1163-1175.

Kellman, L., and C. Hillaire-Marcel, 1998. Nitrate cycling in streams: using natural abundances of NO_3-delta N-15 to measure in-situ denitrification. Biogeochemistry 43: 273-292.

Kendall, C., 1998 Tracing sources and cycling of nitrate in catchments. In Isotope Tracers in Catchment Hydrology, eds. C. Kendall and J. J. McDonnell, pp. 519-576. Elsevier, Amsterdam.

Kendall, C. and R. Aravena, 2000 Nitrate isotopes in groundwater systems. In: Environmental Tracers in Subsurface Hydrology. eds. P.G. Cook and A.L. Herczeg. Kluwer Academic, Boston, pp: 261-297.

Kreitler, C.W. and D.C. Jones, 1975. Natural soil nitrate: the cause of the nitrate contamination of groundwater in Runnels County, Texas., Groundwater, 13: 53-61.

Kreitler, C.W., 1979. Nitrogen isotope ratio studies of soils and groundwater nitrate from alluvial fan aquifers in Texas. J. Hydrology, 42: 147-170.

Letolle, R., 1980. Nitrogen-15 in the natural environment. In: Fritz, P. and Fontes, J.Ch. (Eds) Handbook of Environmental Isotope Geochemistry, v1, Elsevier, New York, pp: 407-433.

Mayer, B., E.W. Boyer, C. Goodale, N.A. Jaworski, N. Van Breemen, R.W. Howarth, S. Seitzinger, G. Billen, L.J. Lajtha, M. Nosal and K. Paustian, 2002. Sources of nitrate in rivers draining sixteen watersheds in the northeastern US: Isotopic constraints Biogeochem., 57: 171-197.

McKinney, R.A., J.L. Lake, M.A. Carpentier and S. Ryba, 2002. Using mussel isotope ratios to assess anthropogenic nitrogen inputs to freshwater ecosystems. Environ. Monitoring and Assessment, 74: 167-192.

Panno, S.V., K.C. Hackley, H.H. Hwang and W.R. Kelly, 2001. Determination of the sources of nitrate contamination in karst springs using isotopic and chemical indicators. Chemical Geology, 179: 113-128.

Showers, W.J., B Usry, M. Fountain, J.C. Fountain, T. McDade and D. DeMaster, 2005. Nitrate Flux from Ground to Surface Waters Adjacent to the Neuse River Waste Water Treatment Plant. Univ. of North Carolina -WRRI Report #351a, 38 pp.

Smil, V., 2002. Nitrogen and food production: proteins for human diets. Ambio, 31: 126-131.

Spalding, R.F. and M.E. Exner, 1993. Occurrence of nitrate in groundwater: a review. J. Environ. Qual., 22: 392-402.

US Department of Agriculture-Natural Resources Conservation Service (USDA NRCS), 1998. 1201, Detailed County Soils-Bladen County, North Carolina: US Dept. of Agriculture-Natural Resources Conservation Service, Raleigh, North Carolina.

USEPA, 1993. Methods for the Determination of Inorganic Substances in Environmental Samples. EPA 600 R-93 100.

van Breemen, N., E.W. Boyer, C.L. Goodale, N.A. Jaworski, K., K. Paustian, S.P. Seitzinger, K. Lajtha, B. Mayer, D. Van Dam, R.W. Howarth, K.J. Nadelhoffer, M. Eve and G. Billen, 2002. Where did all the nitrogen go? Fate of nitrogen inputs to large watersheds in the northeastern USA Biogeochem., 57: 267-293.

Vitousek, P.M., J.D. Aber, R.W. Howarth, G.E. Likens, P.A. Matson, D.W. Schindler, W.H. Schlesinger and D.G. Tilman, 1997, Human alteration of the global nitrogen cycle: Sources and consequences Ecol. Appl., 7: 737-750.

Wassenaar, L., 1995. Evaluation of the origin and fate of nitrate in the Abbotsford Aquifer using the isotopes of ^{15}N and ^{18}O in NO_3-. Applied Geochem., 10: 391-405.