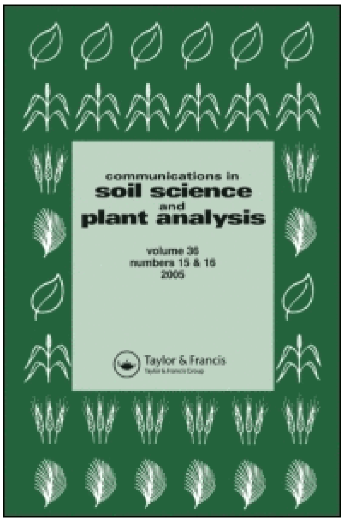

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Denitrification Following Land Application of Swine Waste to Bermudagrass Pasture

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Denitrification Following Land Application of Swine Waste to Bermudagrass Pasture

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Abstract: Nitrogen (N) losses following land-applied animal wastes present an environmental and economical dilemma for producers. Gaseous N losses from pastureland contribute to global warming, ozone depletion, acid rain, and inefficient plant N uptake. This study was designed to monitor nitrous oxide emissions following swine waste and commercial fertilizer treatments to bermudagrass (*Cynedon dactylon* [L.] *pers*) pastures. Denitrification rates were monitored on a biweekly basis for six 0.12-ha bermudagrass pastures for three consecutive growing seasons (1998–2000). Treatments consisted of three split applications of either swine effluent supplemented with ammonium nitrate (SW) or commercial fertilizer (CF). Peak denitrification rates were greatest in 1998, ranging from 0.6 to 1.7 μg N_2O-N m^{-2} h^{-1} for effluent-treated

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plots and 0.3–1.5 μg N_2O-N m^{-2} h^{-1} for commercially fertilized plots. Results from this study suggest denitrification is not a significant N loss pathway in swine waste-amended bermudagrass systems.

Keywords: Denitrification, nitrogen, bermudagrass pasture, land application

INTRODUCTION

Animal manure offers an economical source of nutrient-rich fertilizer and continues to be a major component of agricultural by-products. Currently, swine production in the United States approaches 60,000 million head valued at $76/head (USDA National Agricultural Statistics Service 2001). Southeastern states account for 18% of total swine, with Alabama accounting for 3.6% (USDA National Agricultural Statistics Service 2001). Furthermore, annual production of swine manure in the United States is estimated to approximate 115 billion kg yr^{-1} based on average daily excretion of a 90 kg hog (Midwest Planning Service Committee 1985). Keeping this in mind, mismanagement of N found in animal wastes could lead to substantial N losses prior to plant uptake. Thus, consideration should be given to the accumulation and treatment of swine waste as it relates to atmospheric trace gases.

In addition to supplying plant N, swine effluent also provides facultative anaerobes with a readily available source of carbon (C). Consequently, swine waste amendments stimulate denitrifying bacteria in low-oxygen environments (Nommik 1956; Bremner and Shaw 1958; Limner and Steele 1982; Burford and Bremner 1975; Stanford, Vanderpol, and Dzienia 1975; Ready, Rao, and Jessup 1982). Researchers have shown that denitrification can occur under ideal conditions with an adequate supply of C at temperatures as low as 2–4°C (Nommik 1956; Bremner and Shaw 1958; Limner and Steele 1982; Burford and Bremner 1975; Stanford, Vanderpol, and Dzienia 1975; Ready, Rao, and Jessup 1982). According to Paul and Beauchamp (1989), denitrification rates are highly correlated with concentrations of acetate, butyrate, and propionate in manure. In their study, nine different manures were applied to a poorly drained soil, and consumption of volatile fatty acids (VFAs) was measured. Results showed VFA consumption increased during periods of denitrification. Lynch and Gunn (1978) observed similar results after injecting a soil with 250 mg of acetate-C L^{-1}.

Denitrification remains an important biogeochemical process because it balances N fixation by recycling N to the atmosphere. However, denitrification contributes to gaseous losses of N in the form of nitrous oxide (N_2O-N), nitric oxide (NO), and N gas (N_2), leading to stratospheric ozone destruction, global warming, and loss of a valuable plant nutrient (Crutzen 1974). Nitric oxide emissions also contribute to tropospheric ozone and are one source of acid rain (Galloway and Likens 1981; Logan et al. 1981). Recent field data indicate measurable N_2O-N emissions following swine slurry applications begin almost immediately and return to background rates by day 6 (Maag 1990). Yet, complex spatial and temporal dynamics complicate accurate quantification of denitrification rates (Tiedje, Simpkins, and Groffman 1989). This study was designed to evaluate rate and mass of N loss via denitrification from a swine waste-amended bermudagrass pasture.

MATERIALS AND METHODS

Treatments

In 1998, six 0.12-ha bermudagrass plots were established at a site near Auburn, Alabama (32.41° N, 85.30° W) on a Hiwassee sandy loam (fine, kaolinitic, thermic Typic Rhodudult). Treatments were arranged in a completely randomized design consisting of three split applications of either swine effluent supplemented with ammonium nitrate (SW) or commercial fertilizer applications of ammonium-nitrate (CF) to meet crop requirements of 112 kg N ha^{-1} application $^{-1}$. Swine effluent amendments were determined on the basis of the total phosphorus (P) content of the waste with a target of 9.2 kg P ha^{-1} application^{-1}. Coincident with initial SW amendments, commercially fertilized plots also received a blanket application of triple superphosphate and muriate of potash based on Auburn University Soil Testing Laboratory results. Because of limited effluent storage units in 1998, each application took place over a series of days. Additional storage tanks acquired during the 1999 and 2000 growing season facilitated the effluent application process.

Swine Effluent

Swine effluent was taken from the primary lagoon at the Swine Nutrition Unit at Auburn University. The Swine Nutrition Unit is a 60-sow, "farrow to finish" unit. Animal waste treatment consisted of flushing the farrow unit with recirculated effluent and redelivery to the lagoon for storage. Effluent used for land application was pumped from the primary lagoon at 46 cm below the lagoon surface, avoiding bottom sludge. In the field, a sprinkler system applied an even distribution of effluent during each treatment period. Bermudagrass pasture was cut to 10-cm height and removed prior to each application.

Swine effluent characterization consisted of pH, NH_4-N, NO_3-N, total P, total N, and total suspended solids concentration (Table 1). Nitrate and NH_4-N concentrations were determined colorimetrically (Hue and Evans 1986). Total N was quantified via distillation (Tecator Model 1030 Auto Analyzer) following sulfuric acid dilution and digestion at 400°C according to the procedures listed by Cleseri et al. (Cleseri, Greenberg, and Trussel 1992). Prior

Table 1. Total N (TN), NH_4-N, NO_3-N, P, and pH of swine effluent applied at the study site north of Auburn, Alabama

Date	TN (kg ha^{-1})	NH_4-N (kg ha^{-1})	NO_3-N (kg ha^{-1})	P (kg ha^{-1})	pH (kg ha^{-1})
May 98	59 (6)	42.9 (5.2)	9.9 (4.7)	17.5 (4.2)	7.64 (0.07)
July 98	112 (6)	78.9 (5.2)	22.2 (15.2)	38.4 (4.0)	8.33 (0.07)
August 98	47 (7)	31 (4.7)	9.9 (22.7)	15.3 (2.2)	8.43 (0.07)
April 99	42 (7)	11.8 (0.3)	5.1 (1.2)	6.4 (1.0)	8.05 (0.07)
July 99	43 (9)	11.4 (1.6)	2.2 (.5)	9.8 (2.1)	8.16 (0.24)
September 99	62 (9)	16.5 (1.6)	2.8 (.5)	16.1 (3.6)	8.34 (0.03)
April 00	52 (2)	3.3 (1.4)	17.7 (2.9)	13.0 (2.4)	9.17 (0.10)
June 00	23 (5)	5.4 (0.9)	4.8 (1.)	7.0 (0.7)	8.22 (0.10)
September 00	37 (9)	10.7 (1.9)	11.1 (3.6)	11.8 (2.1)	8.09 (0.10)

Values in parenthesis indicate standard deviation. Note supplemental commercial fertilizer was added to ensure plant N requirements of 112 kg ha^{-1} were met.

to determining P content via inductively coupled plasma spectroscopy, swine waste was digested with a 70:30 nitric to perchloric acid mixture at 200°C (Cleseri, Greenberg, and Trussel 1989). Total suspended solids were determined by using the procedure of Cleseri, Greenberg, and Trussel (1992). Electrical conductivity (Bausch and Lomb Spectronic 100) and waste pH (Accumet pH meter 925) were also measured during each application.

Denitrification

Determination of denitrification consisted of a gas sample collection on a biweekly basis beginning 2 weeks after initial CF or SW applications and ending April 2001. Quantification of denitrification rates in the field was accomplished by using an acetylene inhibition method designed to prevent the conversion of N_2O to N_2 (Tiedje, Simpkins, and Groffman 1989). Thirty-centimeter-long polyvinyl cores (5.8-cm diameter) were used to obtain three soil samples (0–15 cm) per plot with 15 cm of headspace. Cores were subsequently sealed and returned to the ground, leaving half the liner above ground. A syringe was used to remove 60 mL of air (via a septum) from each core, inject 60 mL of acetylene and mix. Headspace samples (3 mL) were collected at 2 and 5 h following acetylene injection. Headspace samples were collected by first injecting 3 mL of air and mixing with a 60 mL syringe. Next, 3 mL of headspace was removed, placed in a sealed glass vial, and subsequently stored at 4°C until analyzed on a gas chromatograph (GC). The GC was equipped with a ^{63}Ni electron capture detector, which operated at 350°C (AllTech Associates, Deerfield, IL). Dinitrogen served as the gas carrier, and N_2O was captured at 50°C using a 3.6-m

Porapak Q 80/100 column. Gaseous N_2O-N concentrations (μL N_2O-N L^{-1}) were determined on the basis of concentration data and gas evolution time. Total N losses via denitrification were quantified by plotting each data point and integrating the area under the curve. Soil samples (0–5 cm) from each plot were analyzed for NO_3-N concentrations and gravimetric soil water content (SWC) coincident with gas sample collection.

Statistics

The Statistical Analysis System (SAS) was used to evaluate denitrification losses from SW and CF bermudagrass plots. Statistical analyses included analysis of variance, Pearson's correlation coefficients, and stepwise linear regression analyses. Prior to regression, SWC and NO_3-N content terms were squared to satisfy statistical assumptions of equal variances. Results were considered significant at the 0.10 level.

RESULTS AND DISCUSSION

Denitrification following SW and CF amendments was greatest during the 1998 growing season, ranging from 0.6 to 1.7 μg N_2O-N m^{-2} h^{-1} on plots treated with SW and 0.3 to 1.5 μg N_2O-N m^{-2} h^{-1} for CF treated plots. Throughout the entire study period, significantly ($p < 0.10$) higher denitrification rates from SW amended plots were observed only during the September 1998 sampling period (Figure 1). Treatment differences at that time were likely a result of meeting total N requirements without the need for supplemental CF. Although an attempt was made to estimate total nutrient content of SW prior to application, actual N and P contents varied. Previous studies have shown that swine waste promotes facultative anaerobic activity via a ready supply of volatile fatty acids (Paul and Beauchamp 1989; Lynch and Gunn 1978). Thus, greater peak denitrification rates following the July 1998 application were not unexpected. During the 1999 and 2000 growing seasons, a proportionate amount of SW and CF, similar to applications 1 and 3 in 1998, was achieved. As a result, denitrification rates from SW-amended plots were much lower than rates observed during the 1998 growing season ranging from 0.4 to 0.6 μg N_2O-N m^{-2} h^{-1}.

Denitrification rates in this study were fairly low compared with previous research. Marshall et al. (1999) found additions of poultry manure resulted in denitrification rates ranging from -20 to 2500 μg N m^{-2} h^{-1}. In this study, observations correspond more closely with those of Lowrance et al. (1998) who report average denitrification rates of 0.1–2.2 μg N m^{-2} h^{-1}. Seasonal variation was also observed, showing latent peaks in N_2O-N emissions during the 1998 and 1999 growing seasons from May to September. As the season progressed, denitrification rates did not exceed 0.10 μg N m^{-2} h^{-1}.

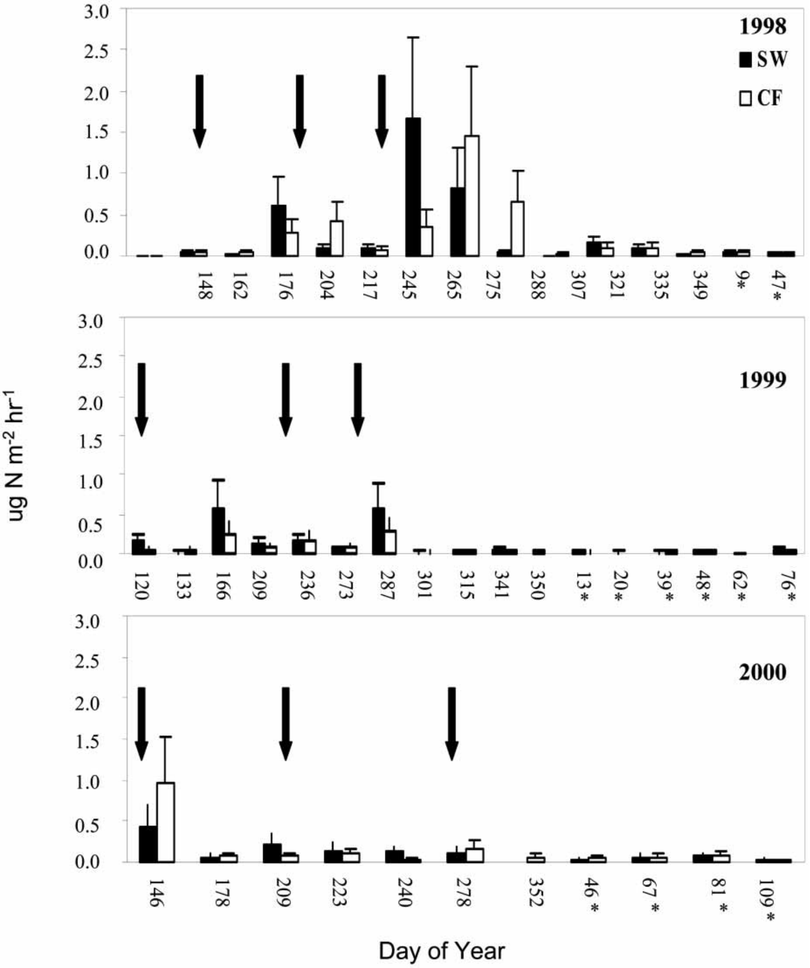

Figure 1. Denitrification rates following applications of swine effluent supplemented with ammonium nitrate (SW) or commercial fertilizer (CF) at the study site near Auburn, Alabama for the 1998–2000 measurement period. Arrows denote time of amendment application. *Denotes a new calendar year.

Denitrification rates varied within both SW amended and CF plots throughout the 3-year study. Variation in N_2O emissions reflects the many spatial and temporal characteristics involved in the process of denitrification. Large variations in measured denitrification rates have been well documented and are not uncommon (Lessard et al. 1996; Cabrera and Chiang 1994; Allen, Jarvis, and Headon 1996; Burton and Beauchamp 1985; Parkin and Meisinger 1989). Beauchamp, Bergstrom, and Burton (1996) was unable to correlate N_2O production reliably with crop type, management, or soil conditions in a field study using an acetylene core method with various cropping systems and soil amendments. Under ideal field conditions, facultative anaerobes control rates of denitrification, and consideration of environmental factors influencing these organisms may aid in understanding the extent and variability associated with denitrification.

Research indicates a strong relationship exists between denitrification, water-filled porosity, and CO_2 (Parsons, Murray, and Smith 1991). Others suggest that peak N_2O fluxes correlate well with NO_3-N concentrations in the soil (Lowrance et al. 1998; Lessard et al. 1996; Cabrera and Chiang 1994). In this study, stepwise regression with SWC and NO_3-N content was significant linearly related to observed denitrification rates (Table 2). Results indicated that at SWCs greater than 12%, regression with SWC alone explained 59–71% of the variability in denitrification rates. On two occasions, a combination of SWC and NO_3-N content was used to explain 94–97% of the variability in denitrification rates. However, in both cases, SWC content accounted for greater than 65% of the total variability. Results illustrate that gravimetric SWC was the predominant variable controlling denitrification rates. Nitrate-N concentrations were not significantly different between treatments during much of the measurement period and not limiting during periods of peak denitrification (Figure 2).

Keeping this in mind, soil NO_3-N concentration and SWC did not reliably account for the variability in denitrification between peaks. Reasons for the

Table 2. Stepwise linear regression parameters relating soil water (SWC) and NO_3-N contents to denitrification rates

Date	Variable	Slope	Intercept	Soil water content %	Soil NO_3-N kg ha^{-1}	r^2
06/25/98	SWC	−1.88	0.032	7.79 (0.01)		0.51
09/02/98	NO_3-N	2.42 E-7	0.05	10.81 (0.01)		0.85
05/13/99	SWC	−2.53				
	NO_3-N	3.16	0.08	14.08 (0.01)	80.05 (24.51)	0.94
06/15/99	SWC	18.09	−0.22	18.38 (0.02)		0.71
09/30/99	SWC	5.98	−0.63	14.73 (0.01)		0.57
10/28/99	SWC	−2.36	0.039	11.79 (0.02)		0.63
1/13/00	SWC	1.24	−0.02	18.77 (0.01)		0.57
02/17/00	NO_3-N	4.01 E-7	0.02		22.09 (0.01)	0.87
07/28/00	SWC	−31.09				
	NO_3-N	−4.21 E-6	0.81	13.57 (0.00)	141.05 (21.44)	0.97
02/15/01	SWC	0.17	−3.52	18.58 (0.01)	0.59	

Soil water and NO_3-N contents are given with standard errors in parentheses. All results are significant at alpha = 0.10.

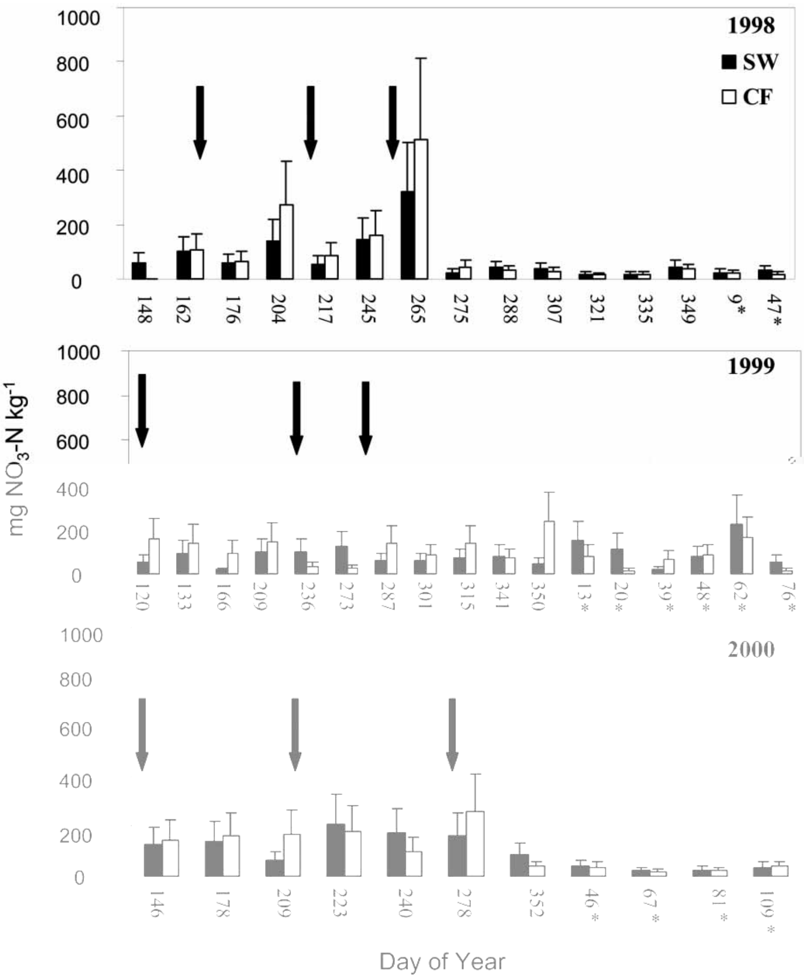

Figure 2. Surface soil (0–5 cm) NO_3-N concentrations following applications of swine effluent supplemented with ammonium nitrate (SW) or commercial fertilizer (CF) at the study site near Auburn, Alabama for the 1998–2000 measurement period. Arrows denote time of amendment application. *Denotes a new calendar year.

inconsistency in our ability to predict denitrification rates are likely a function of microbial activity and microclimate, because no single pattern of NO_3-N and SWC could be used to explain peak denitrification rates. Lessard et al. (1996) suggested poor correlations between N_2O flux within the soil and at the soil surface indicated restricted gas flux. Previous studies have found anaerobic microsites in fine textured soils may be present even at low water contents and impact observed variations in denitrification (Loro, Bergstrom, and Beauchamp 1997; Groffman and Tiedje 1991). Manure changes microbial activity and necessarily affects denitrification rates. Lessard et al. (1996) found contributions of C and N from manure altered soil physical and chemical properties and ultimately influenced microbial activity and thereby N mineralization. Although microbial effects relative to C : N ratios of waste were not evaluated in this study, it is well established VFAs present in swine waste stimulate microbes, consequently depleting soil O_2 and increasing denitrification rates (Nommik 1956; Bremner and Shaw 1958; Ready, Rao, and Jessup 1982; Paul and Beauchamp 1989; Papendick and Campbell 1980). As C becomes increasingly available, N becomes the rate-limiting nutrient in denitrification (Kohl et al. 1976), thus availability of N over time may explain some temporal variation.

Nitrous oxide emissions from pastureland comprise 10.0% of total N_2O-N flux and contribute to global warming and photolytic depletion of stratospheric ozone (Beauchamp, Bergstrom, and Burton 1996). Total denitrification losses during the 1998 and 1999 measurement periods were significantly greater from SW amended plots ($p < 0.10$), resulting in 2–4.5 kg N_2O-N ha^{-1} compared to 1.3–2 kg N_2O-N ha^{-1} from CF treatments (Figure 3). Nitrogen losses expressed as a fraction of total N applied may be a more practical representation of agronomic and environmental impacts as a result of denitrification from bermudagrass pasture. Total N losses ranged from 0.6 to 1.3% and 0.4 to 0.6% of N applied as SW or CF, respectively (Figure 3). Thus, despite greater N losses from SW amended plots, denitrification does not represent a substantial N loss pathway in swine effluent-amended systems.

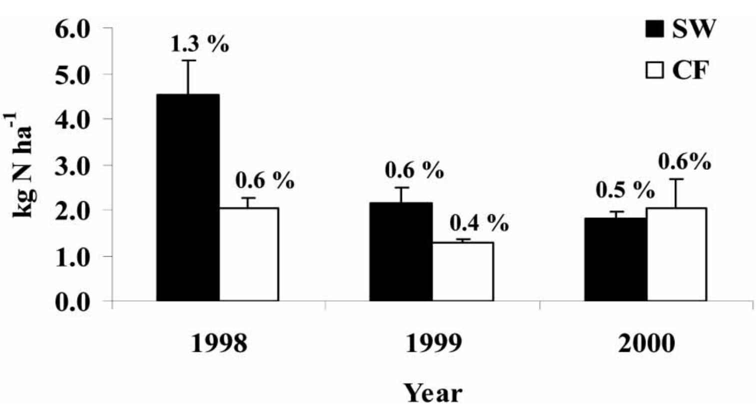

Figure 3. Cumulative yearly N_2O-N losses via denitrification following applications of swine effluent supplemented with ammonium nitrate (SW) or commercial fertilizer (CF) at the study site near Auburn, Alabama for the 1998, 1999, and 2000 measurement periods. Values listed above each bar represent N losses as a percentage of the total N applied each year.

CONCLUSIONS

Denitrification losses from SW- and CF- amended plots varied throughout the measurement period. Peak periods of N_2O-N emission were best explained via SWC and soil NO_3-N content in stepwise regression. Yet variability in observed denitrification rates during the remainder of the measurement period was not well explained by either SWC or NO_3-N content. Results indicate that although SWC or NO_3-N content is oftentimes the most limiting factor, microbial activity and microclimate also impact observed denitrification rates.

In this study, latent peaks of denitrification occurred mostly during the growing season following application of SW or CF. Although few significant differences in peak denitrification rates were observed between SW and CF, it should be noted that denitrification from SW-amended plots during these periods was typically higher than CF treatments. As a result of those differences, total N losses throughout the measurement period were significantly higher from SW-amended plots resulting in 1–2.5 kg N ha^{-1} greater N losses each year. However, total N losses did not exceed 2% of N applied as either SW or CF. These data suggest minimal agronomic and environmental impacts via gaseous losses of N_2O-N from SW-amended bermudagrass in southeastern United States.

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