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Research Article

Tillage and Fertilizer Management Effects on Soil-Atmospheric Exchanges of Methane and Nitrous Oxide in a Corn Production System

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Land application of poultry litter (PL) presents an opportunity to improve soil productivity and disposal of poultry waste. We investigated methane (CH_4) and nitrous oxide (N_2O) emissions from agricultural soil receiving PL and ammonium nitrate (AN) fertilizers using surface (SA), soil incorporation (SI), and subsurface band (BA) application methods in conventional (CT) and no-tillage (NT) systems on a Decatur silt loam soil in North Alabama. Plots under CT and NT were sinks of CH_4 in spring, summer, and fall. In winter, the plots had net emissions of 3.32 and 4.24 g CH_4 ha^{-1} day^{-1} in CT and NT systems, respectively. Plots which received AN were net emitters of CH_4 and N_2O, whereas plots which received PL were net sinks of CH_4. Plots which received PL using SA or SI methods were net emitters of N_2O, whereas under PL using BA application, the plots were net sinks of N_2O. Our study indicates that using subsurface band application of PL was the most promising environmentally sustainable poultry waste application method for reducing CH_4 and N_2O emissions from agricultural soil in NT and CT corn production systems on the Decatur soil in north Alabama.

1. Introduction

The presence of the poultry industry in close proximity to row crop farming systems in north-eastern and north-central Alabama presents an opportunity to economically improve crop yields and soil quality through land application of the carbon and nitrogen-rich poultry litter (PL). Poultry litter is an organic fertilizer which is a valuable source of plant nutrients and an excellent soil amendment for improving soil quality and productivity. In addition to exploiting its economic benefits as a fertilizer, application of poultry litter to agricultural soils in conservation tillage systems is a recommended method for disposing of the large quantities of litter generated in the poultry industry of the southeastern USA [1–3].

The increasing size and concentration of animal production units has given rise to concerns about air emissions on the earth's atmosphere at local and global scales [4]. Methane (CH_4) and nitrous oxide (N_2O) are major agricultural greenhouse gases with greater global warming potential than carbon dioxide (CO_2) and can significantly contribute to climate change. According to the Intergovernmental Panel on Climatic Change, CH_4 is about 20 to 21 times more effective as a greenhouse gas than CO_2, while N_2O has a direct global warming potential 170 to 290 times that of CO_2 [5].

Agricultural soil is a natural source of CH_4 and N_2O greenhouse gases. Soil microbial activity is the primary factor leading to the production of biogenic gases such as CH_4 and N_2O [6, 7]. Soil CH_4 emission is a product of CH_4 oxidation and methanogenesis [8]. According to Conrad [9], the two microbial processes of CH_4 oxidation and methanogenesis can occur simultaneously, even in arable terrestrial ecosystems. The three groups of organisms which take part in the microbial processes leading to soil CH_4 fluxes are methanotrophic bacteria, ammonia oxidizing bacteria, and methanogenic bacteria [10]. The majority of soil N_2O emissions come from N_2O produced as an intermediate during nitrification and denitrification processes [11]. Therefore, soil and animal waste management strategies which can reduce or prohibit CH_4 formation and inhibit soil denitrification process can minimize emissions of CH_4 and N_2O greenhouse gases from agricultural soils.

Use of organic and inorganic fertilizers for crop production has been widely documented to increase emissions of agricultural greenhouse gases [12–17]. Manures from poultry and livestock contain proteins, amino acids, and carbohydrates, which provide a source of energy for bacteria, which, upon decomposition, can release greenhouse gases such as CO_2 and CH_4, in addition to odor causing gases such as ammonia (NH_3) and hydrogen sulfide (H_2S). Odors, gases, and particulate matter from use of animal manures in agriculture are a cause of concern from social, environmental quality, and human health points of view. Land application of animal waste has been reported to increase soil emissions of greenhouse gases [18, 19]. Therefore, if not properly managed, soil-applied PL can potentially contribute to the enrichment of the atmosphere with greenhouse gases such as CH_4 and N_2O.

Due to their relatively long chemical lifetime and stability, CH_4 and N_2O can be transported into the stratosphere where they contribute to destruction of ozone [16, 20]. Consequently, while land application of PL presents an opportunity for disposing of the large quantities of poultry waste, there is a need to develop soil, crop, and animal utilization strategies to minimize emissions of greenhouse gases from agricultural soil which can make the disposal of poultry litter through land application environmentally sustainable. In areas of intensive animal production with limited disposal areas such as the intensive poultry producing region of north Alabama, repeated applications of manure on the same piece of land are common. This not only increases the risk of soil and water pollution from excess nutrients such as N [21] and P but also can potentially lead to increased soil emissions of agricultural greenhouse gases.

In addition to being a source of soil CH_4, agricultural soils can be effective sinks of atmospheric CH_4, which can have a significantly impact on net ecosystem CH_4 balances. Agricultural practices such as tillage and fertilizer management practices can affect the soil's ability to act as a sink of atmospheric CH_4 [22]. Studies on greenhouse gas emissions from animal waste-treated soils have largely dealt with dairy cattle or liquid swine manure in the Midwestern USA and elsewhere. Information regarding greenhouse gas emissions from soils treated with poultry waste in the Southeastern USA is lacking. Studies to document the impact of soil and fertilizer management strategies on current and future inventories of soil greenhouse gas emissions, taking into account the various types of animal wastes and different agroecological regions, are needed. The objectives of this study were to investigate emissions of CH_4 and N_2O from agricultural soil receiving PL and ammonium nitrate (AN) fertilizers using surface, soil incorporation, and subsurface band application methods in conventional tillage (CT) and no-tillage (NT) systems in a corn (*Zea mays*, L.) production system.

2. Materials and Methods

2.1. Study Location. The study was conducted at the Winfred Thomas Agricultural Experiment Station, Hazel Green, Alabama (latitude 34°89′ N and longitude 86°56′ W), which is strategically situated in a major row crop and poultry producing area in the Tennessee Valley region of north Alabama. The soil at the study site is a Decatur silt loam (fine, kaolinitic, thermic Rhodic Paleudult). We chose a fallow field under tall fescue grass (*Festuca arundinacea*, L.) which had not received organic or inorganic fertilizers for over 10 years to eliminate the effect of previously added soil nutrients on CH_4 and N_2O exchanges in the study plots. Baseline soil properties in the plots prior to treatment establishment are presented in Table 1.

2.2. Treatments and Experimental Design. Treatment factors consisted of two tillage systems: conventional tillage (CT) and no-tillage (NT); two N sources: poultry litter (PL) and ammonium nitrate (AN); and three fertilizer application methods: surface application (SA), soil incorporation (SI), and subsurface band application (BA) (poultry litter only). The fertilizers were applied at a rate of $150\,\mathrm{kg\,N\,ha^{-1}}$ to represent the recommended rate for corn on the Decatur soil type. In addition, a tall fescue grass fallow and a $0\,\mathrm{kg\,N\,ha^{-1}}$ treatment were used to determine background CH_4 and N_2O greenhouse gas emissions in plots without tillage and fertilizer application.

The plots were arranged in a randomized complete block design with four replications. However, gas sampling was done in the first three replications for practical and technical feasibility reasons, such as sampling time and laboratory instrumentation capabilities. Gross plot size was $8\,\mathrm{m} \times 8\,\mathrm{m}$. The replications were separated from each other by a fescue grass strip 8 m wide, while individual plots within a replication were separated from each other by a 2 m wide tall fescue grass strip to hydrologically isolate different treatments from each other and prevent nutrient spill over from plots receiving fertilizer treatments into adjacent plots.

Conventional tillage included fall chisel plowing using a field cultivator and spring disking to prepare the seedbed before corn planting as per local farmer's practice. No-tillage treatment involved planting a wheat (*Triticum aestivum* L.) cover crop in the fall of 2007, 2008, and 2009 and killing the cover crop with glyphosate herbicide prior to planting corn in the standing residue in the spring of each year. The wheat cover crop did not receive any fertilizer to encourage it to efficiently "scavenge" residual soil nutrients and incorporate them as above ground biomass during the winter season to reduce runoff and leaching losses of nutrients. The amount of poultry litter to supply $150\,\mathrm{kg\,N\,ha^{-1}}$ was calculated based on the N content of poultry litter, which was determined using the LECO TruSpec CN analyzer (LECO Corporation, St. Joseph MI).

Table 1: Baseline soil properties at the study site prior to establishing treatments, Hazel Green, Alabama.

Soil depth (cm)	pH	NH_4-N mg kg^{-1}	NO_3-N mg kg^{-1}	PO_4-P mg kg^{-1}	Total C (g kg^{-1})
0–5	6.21	9.22	15.26	6.15	14.22
5–10	6.19	8.15	23.12	5.08	8.53
10–20	6.31	5.64	15.18	6.10	7.04
20–30	6.36	6.15	12.71	4.46	5.09
30–60	6.26	5.81	8.16	2.17	4.03
60–120	6.03	4.65	5.40	1.67	3.70
S.E.	0.2	3.7	8.8	4.4	0.3

Table 2: Tillage and fertilizer treatments used in the CH_4 and N_2O greenhouse gas flux study, Hazel Green, Alabama.

Treatment code	Tillage system	N source	N rate (kg N ha^{-1})	N application method
CT-0N	Conventional (CT)	None	0	N/A
CT-150AN-SA	Conventional (CT)	AN†	150	Surface application (SA)
CT-150AN-SI	Conventional (CT)	AN	150	Soil incorporation (SI)
CT-150PL-SA	Conventional (CT)	PL	150	Surface application (SA)
CT-150PL-BA	Conventional (CT)	PL	150	Subsurface band application (BA)
CT-150PL-SI	Conventional (CT)	PL	150	Soil incorporation (SI)
NT-0N	No-till (NT)	None	0	N/A
NT-150AN-SA	No-till (NT)	AN	150	Surface application (SA)
NT-150PL-SA	No-till (NT)	PL	150	Surface application (SA)
NT-150PL-BA	No-till (NT)	PL	150	Subsurface band application (BA)
GF-0N	Grass Fallow (GF)	None	0	N/A

†AN: ammonium nitrate; PL: poultry litter.

Surface application involved broadcasting weighed quantities of PL or AN fertilizers to supply 150 kg N ha^{-1} on the soil surface in both CT and NT systems. Soil incorporation application method involved broadcasting weighed quantities of PL or AN fertilizers to supply 150 kg N ha^{-1} on the soil surface followed by soil incorporation using a rototiller in CT system.

Subsurface band application was achieved by applying known quantities of PL to supply 150 kg N ha^{-1} in a narrow band, 4.4 cm wide to a depth of about 7 cm and covering it with soil using a prototype implement for subsurface band application of PL developed at the USDA-ARS National Soil Dynamics Laboratory, Auburn AL [23]. The distance between the PL bands was 97 cm. Soil incorporation of fertilizer was not used in the NT system, while subsurface banding was not used with AN fertilizer, resulting in an incomplete factorial set of treatments (Table 2). Field operations for fertilizer application and fertilizer incorporation were done on April 23, 2008 and April 24, 2009. Corn was planted in the plots at 97 cm row spacing on April 24 in spring of 2008 and 2009.

2.3. Gas Sampling and Laboratory Analyses. Soil gas samples were collected using custom-built static PVC chambers designed in accordance with the USDA-ARS GRACEnet Chamber-based Trace Gas Flux Measurement Protocol [24]. The chambers consisted of two parts: a chamber anchor base and a vented sampling chamber head. The chamber bases were made from white PVC pipe, 20 cm inside diameter, 6 mm thick, and 15 cm long. The chamber bases were driven 10 cm into the ground using a rectangular wooden block and a rubberized mallet, leaving a soil collar 5 cm above the ground. Two chamber bases were installed in each plot immediately after corn planting, one between corn rows and one within the corn row. The chamber heads were made from 10 cm long socket end-caps of the same white PVC pipe material which was used to make the anchor bases and are designed to fit closely over the PVC pipe chamber anchor bases. The white external surfaces were chosen to reflect sun rays to prevent temperature increase inside the chambers during gas sampling.

Three tight fitting butyl rubber corks were glued to the top of the flux chamber head. A PVC vent tube 10 cm long and 4.8 mm inside diameter was inserted into the first butyl rubber cork on top of the flux chamber head to offset pressure differences between the inside and outside of the flux chamber during measurements. A thermometer was inserted into the second butyl rubber cork to measure temperature inside the flux chamber during measurements. The third butyl rubber cork was used as a sampling port into which a syringe needle was inserted during gas sampling. In addition, a small fan driven by a 12 V DC electric motor was mounted on the inside wall of the flux chamber head to thoroughly mix the air inside the flux chamber during gas sampling.

The chamber anchor bases were kept open at all times except during gas sampling time. During gas sampling, the chamber anchor bases were fitted with the tight fitting vented chamber heads. Gas samples were collected into 25 mL polypropylene leak-proof syringes with a 0.72 mm outside diameter (22 gauge) flat-end needle. During gas sampling, the air inside the chamber was mixed by repeatedly pumping air into the syringe and expelling the air, three to five times. Gas samples were extracted at 0-, 10-, and 20-minute time intervals. Sampling was typically restricted to between 10 am and 2 pm to avoid large temperature fluctuations during gas collection periods. Gas sampling was done once prior to tillage and planting operations immediately after planting in spring in 2008 and 2009. Post-planting gas sampling was done at a frequency of at least once every month during spring, summer, fall, and winter seasons in each year.

Immediately after extraction, the 25 mL gas samples were transferred to 20 mL pre-evacuated vials sealed with butyl rubber septa (Varian Inc, Palo Alto, CA), which over-pressurized the system to ensure that in case of any leakage, gas sample movement will only be out of the vials and not into the vials. The vials containing the soil gas samples were put into Ziploc bags and taken to the laboratory for greenhouse gas analyses. The gas samples were analyzed using a Varian CP-3800 gas chromatograph (Varian Inc, Palo Alto, CA) equipped with a ^{65}Ni electron capture detector (ECD) and a flame ion detector (FID) for N_2O and CH_4 concentration measurements, respectively.

2.4. Surface Flux Calculation. Surface fluxes of CH_4 and N_2O were calculated using the following equation by Hutchinson and Livingston [25]:

$$fo = \frac{\Delta C}{\Delta T} \times \frac{V}{A} \times \frac{M}{V_{\mathrm{mol}}}, \tag{1}$$

where fo is the flux rate of soil CH_4 or N_2O gas (μg m^{-2} min^{-1}), $\Delta C/\Delta T$ is the rate of change of gas concentration inside the measuring chamber (μg min^{-1}), V is the head-space volume of the measuring chamber (m^3), A is the surface area of the measuring chamber (m^2), M is the molecular weight of the gas (16 and 44 g mol^{-1} for CH_4 and N_2O resp.), and V_{mol} is the molar volume of gas (m^3 mol^{-1}). In each plot, flux values from the flux chamber within the crop row and the flux chamber between the crop row were averaged to represent the flux value for each plot.

In plots with the subsurface band PL application treatment, the effective flux for the plot was calculated taking into account that the PL band covered only a narrow (4.4 cm wide) strip within the chamber. The adjustment for the effective soil gas flux from the PL band was done using the equation by Way et al. [26]:

$$F_{E,B} = \frac{(F_B W_B + \mathrm{F}_{\mathrm{FC,Ctrl}}(S_B - W_B))}{S_B}, \tag{2}$$

where $F_{E,B}$ is the effective soil gas flux from a banded plot (μg m^{-2} min^{-1}); F_B is the soil gas flux from the band alone (μg m^{-2} min^{-1}); W_B is the width of the PL band (m); $F_{\mathrm{FC,Ctrl}}$ is the soil gas flux from the chamber without a PL band (μg m^{-2} min^{-1}); S_B is the center-to-center band spacing (m). This adjustment was done to account for the fact that greenhouse gas fluxes per unit area directly above the PL band are different from fluxes from the area inside the same chamber where there is no band. It also accounts for the fact that the PL bands do not actually cover the whole plot area.

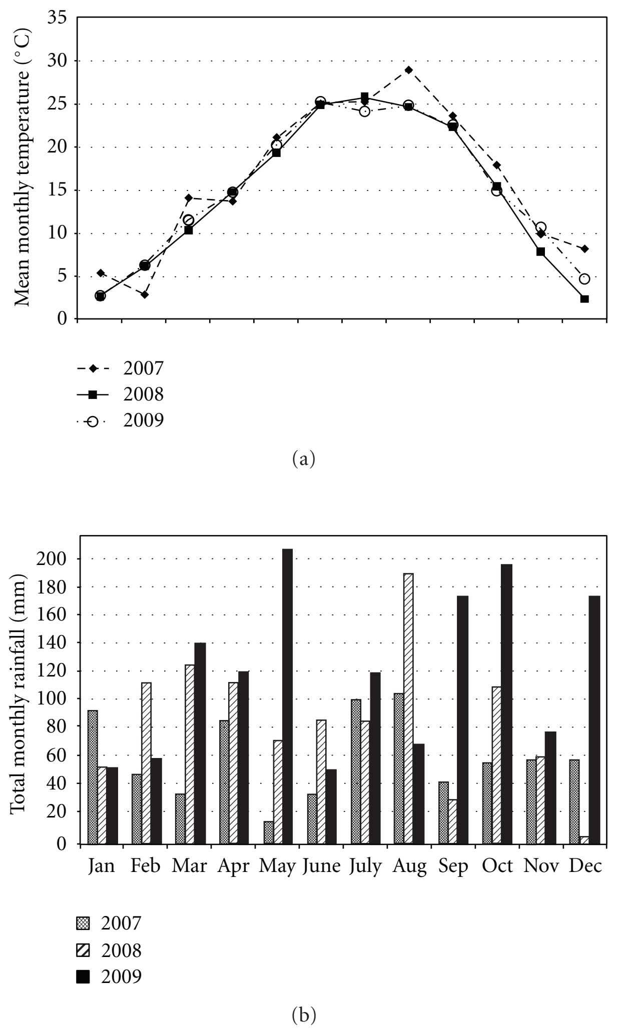

Figure 1: Monthly rainfall totals and mean monthly temperatures at study site, Hazel Green, Alabama.

2.5. Ancillary Data. During gas sampling, soil temperature in the top 5–10 cm of the soil was measured using Mannix digital soil thermometers permanently installed in the plots. Daily weather data were recorded using an automatic weather station at the study site. Mean monthly temperature and total monthly rainfall at the study site during the study period are presented in Figure 1.

2.6. Statistical Analyses. The general linear models analysis of variance (ANOVA) procedure [27] was used for analyzing the data collected in this study using SAS, ver. 9.03 software [28].

The least significant difference test was used to compare treatment mean differences for the measured variables. Pearson correlation analysis was used to determine relationships between CH_4 and N_2O greenhouse gas fluxes and other measured variables such as soil temperature.

Table 3: Soil methane fluxes as influenced by tillage systems and ammonium nitrate (AN) and poultry litter (PL) N fertilizers using surface (SA), soil incorporation (SI), and subsurface band (BA) application methods in spring, summer, fall, and winter seasons, Hazel Green, Alabama.

	Spring	Summer	Fall	Winter	Mean flux
			CH_4 flux ($g\ ha^{-1}\ day^{-1}$)		
Tillage systems					
Conventional tillage	−2.05a†	−1.96ab	−0.68a	3.32b	−0.34
No-tillage	−2.23a	−2.99a	−1.94a	4.24b	−0.73
Grass fallow	−1.79a	−0.26b	2.38b	−21.44a	−5.28
Fertilizer treatments					
0N control	−5.97a	−1.37b	0.60b	22.15d	3.85
150 kg N ha^{-1} AN-SA	−5.48a	−0.44c	−0.87b	32.16e	6.34
150 kg N ha^{-1} AN-SI	3.49bc	−5.74a	−1.21b	4.65c	0.29
150 kg N ha^{-1} PL-SA	4.16c	−0.30c	−1.09b	−13.12b	−2.58
150 kg N ha^{-1} PL-SI	0.32b	−2.96b	1.62b	−13.95b	−3.74
150 kg N ha^{-1} PL-BA	−5.25a	−5.41a	−4.77a	−18.00a	−8.36
Seasonal means	−1.74	−2.22	−0.52	−1.16	

† Means for tillage system or N fertilizer treatment within a season followed by the same letter are not significantly different at the $P \leq 0.05$ level.

3. Results and Discussion

3.1. Methane (CH_4) Flux. The trend in soil CH_4 fluxes for the 11 treatments at different sampling periods during the two-year study period shows variations in CH_4 fluxes within and among treatments at different sampling periods (Figure 2). Means of soil CH_4 fluxes for tillage and N fertilizer treatments in spring, summer, fall, and winter during the study period are given in Table 3. Soil CH_4 fluxes in the plots under CT and NT systems were negative in spring, summer, and fall, whereas in winter, positive soil CH_4 fluxes of 3.32 and 4.24 g CH_4 ha^{-1} day^{-1}were observed in plots under conventional and NT systems, respectively (Table 3).

A positive gas flux implies that the measured gas was moving from the soil to the atmosphere during the time of measurement. This happens when the value of the term "$\Delta C/\Delta T$" in the equation $fo = \Delta C/\Delta T \times V/A \times M/V_{\text{mol}}$ is positive; that is, the gas concentration inside the sampling chamber during the measuring time was increasing with time. A negative gas flux is the reverse of the above, which implies that the gas was moving from the atmosphere into the soil during the time of measurement; that is, the concentration of gas inside the chamber was decreasing with time during the time of measurement. In that case, the term "$\Delta C/\Delta T$" in the equation $fo = \Delta C/\Delta T \times V/A \times M/V_{\text{mol}}$ is negative.

The above results show that plots under CT and NT systems were sinks of atmospheric CH_4 in spring, summer, and fall. However, the same plots were emitters of soil CH_4 in winter. Plots under the grass fallow system were sinks of atmospheric CH_4 in spring, summer, and winter, whereas in fall, the plots were emitters of CH_4. The seasonal variation in soil CH_4 fluxes among tillage systems was attributed to changes in soil environmental conditions. Soil CH_4 fluxes were significantly positively correlated ($P < 0.03$) to soil moisture content and negatively correlated to soil temperature ($P < 0.02$) at sampling time. In spring and summer, the soil acted as a sink of atmospheric CH_4 in CT and NT systems which had lower soil moisture content, and less so in grass fallow plots which had higher soil moisture content. In winter when CT and NT plots had relatively higher soil moisture content, the plots were net emitters of CH_4 (Table 3).

According to Linn and Doran [29], the percentage of soil pore space filled with water, as determined by water content, and total porosity appears to be closely related to soil microbial activity under different tillage regimes. The fact that even for the same soil type, soil microbial activity under CT, NT, and grass fallow systems can vary due to different soil moisture content can explain the differences in soil CH_4 fluxes between different tillage systems. A greater pore continuity and the presence of ecological niches for methanotrophic bacteria in NT systems lead to increased CH_4 uptake compared to CT systems.

Plots which received N fertilizer in the form of inorganic AN fertilizer using surface application method (AN-SA) or PL using the subsurface band application method (PL-BA) and the control (0 kg N ha^{-1}) plots had negative soil CH_4 fluxes of about −5.00 g CH_4 ha^{-1} day^{-1} in spring (Table 3). Plots which received 150 kg N ha^{-1} AN using soil incorporation application method (AN-SI), PL using surface application method (PL-SA), and PL using soil incorporation method (PL-SI) had mean soil CH_4 emissions ranging from 0.32 to 4.16 g CH_4 ha^{-1} day^{-1} in spring. In summer and fall, soil CH_4 fluxes in the plots were generally negative, irrespective of N fertilizer treatment. However, in winter, plots which received AN-SA, AN-SI, and the control plots had mean soil CH_4 emissions ranging from 4.65 to 32.16 g CH_4 ha^{-1} day^{-1}, while plots which received PL-SA, PL-SI, and PL-BA had negative soil CH_4 fluxes ranging from −18.00 to −13.12 g CH_4 ha^{-1} day^{-1} (Table 3). Seasonal soil CH_4 fluxes averaged over tillage and fertilizer treatments show that the plots were sinks of CH_4 in all seasons.

The data for soil CH_4 fluxes at various sampling periods during the study period as shown in Figure 2 is useful in

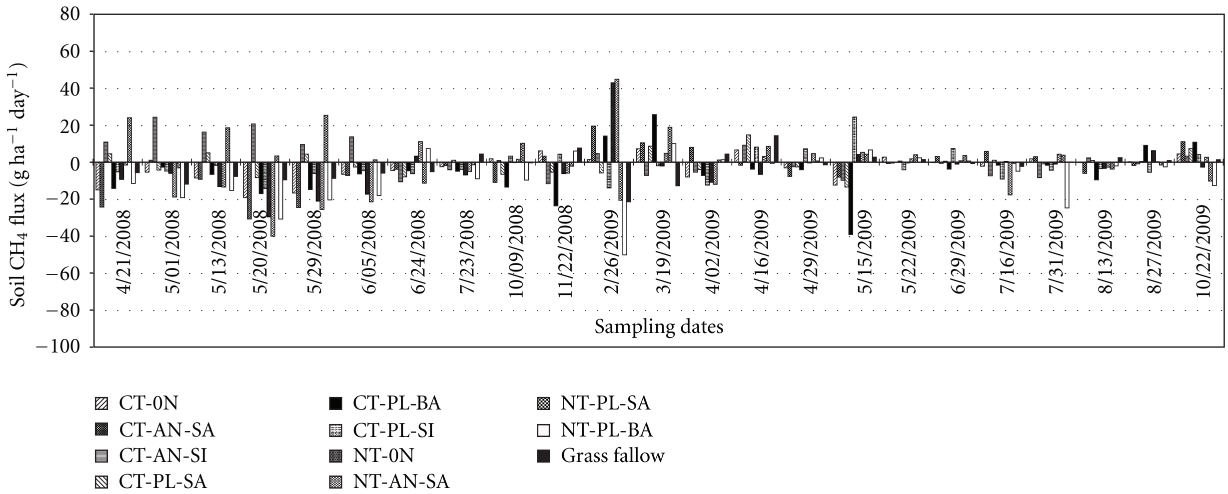

Figure 2: Soil CH_4 fluxes in corn plots receiving ammonium nitrate (AN) and poultry litter (PL) N fertilizers using surface (SA), soil incorporation (SI), and subsurface band (BA) application methods in conventional tillage (CT) and no-tillage (NT), systems, Hazel Green, Alabama.

showing trends in gas fluxes over time. The different soil management practices represented by tillage and fertilizer treatments show a pattern whereby the soil fluctuated between being a source and a sink of CH_4. Therefore, mean CH_4 flux data over time is more important in quantifying the net effect of different crop and soil management strategies and hence in making assessments of agricultural contributions of greenhouse gases to the environment. Mean CH_4 fluxes in each tillage system averaged over seasons during the study period were −0.34, −0.73, and −5.28 g CH_4 ha^{-1} day^{-1} in plots under CT, NT, and grass fallow plots, respectively (Table 3).

These results show that all the plots used in the study were net sinks of atmospheric CH_4 irrespective of tillage system. This is consistent with reports by other researchers, that in general, upland soils are CH_4 sinks [30]. In agreement with our findings, Johnson et al. [31] also found no significant differences in soil CH_4 fluxes between CT and NT. However, Alluvione et al. [32] found higher CH_4 emissions in NT (20.2 g CH_4 ha^{-1}) compared to CT (1.2 g CH_4 ha^{-1}) on a clay loam soil in northeastern Colorado. Since upland terrestrial soils are generally CH_4sinks, they can play a significant role in mitigating the enrichment of the atmosphere with this greenhouse gas. In our study, the grass fallow plots were the highest sinks of atmospheric CH_4. Several other studies have also shown that grassland soils are usually sinks of CH_4 [33–37].

Mean CH_4 fluxes in each N fertilizer treatment averaged over seasons during the study period show that plots which received inorganic AN fertilizer and the control plots were net emitters of soil CH_4 irrespective of application method, with fluxes ranging from 0.29 to 6.34 g CH_4 ha^{-1} day^{-1}, whereas plots which received N fertilizer in the form of PL were net sinks of atmospheric CH_4 irrespective of application method, with fluxes ranging from −8.36 to −2.58 g CH_4 ha^{-1} day^{-1} (Table 3). Mean CH_4 fluxes averaged over all the treatments during the study show that the plots were a net sink of CH_4 with a flux of −5.64 g CH_4 ha^{-1} day^{-1}.

In soils, CH_4 is produced from anaerobic decomposition of organic material by methanogenic bacteria which consume soil CO_2 and convert it to CH_4 gas. According to R.S. Hanson and T.E. Hanson [38], oxidation of CH_4 by methanotrophic bacteria is the only known net biological sink for atmospheric CH_4 and terrestrial emissions. Subsurface band application of PL can cause an increased rate of soil microbial activity under warm temperatures and moist soil conditions in aerated soils in spring and summer. Under favorable soil conditions, methanotrophic soil bacteria consume CH_4 gas in the soil which favors soil CH_4 uptake. Nitrogen fertilization has been demonstrated to stimulate certain soil methanotrophs, while inhibiting other methanotrophs [39]. Therefore, increased microbial activity can result in increased rate of CH_4 consumption by methanotrophic bacteria which may have a net effect of causing the soil to be a sink of atmospheric CH_4.

According to Bédard and Knowles [40], oxidation of CH_4 in soils can also be carried out by nitrifying bacteria, which have an affinity for CH_4 similar to that of the common methanotrophs. This may suggest that conditions favorable for nitrification bacteria, such as the supply of C and N rich organic matter from PL, can actually promote the net movement of atmospheric CH_4 into the soil when the PL is concentrated in a band. On the other hand, inorganic N fertilization has been reported to inhibit CH_4 consumption activity in arable soils [41–44]. This may explain why plots which received 150 kg N ha^{-1} in the form of inorganic AN fertilizer had cumulative CH_4 emissions whereas plots which received 150 kg N ha^{-1} in form of PL were net sinks of CH_4 (Table 2). In a tropical rice field alluvial soil, Nayak et al. [45] reported that organic and mineral fertilizer stimulated CH_4 oxidation whereas application of both organic and mineral fertilizer inhibited CH_4 oxidation probably due to N immobilization. In contrast to most findings, Hernandez-Ramirez et al. [46] reported that soils which received manure in a continuous corn cropping system were net seasonal CH_4 emitters, whereas similar plots receiving inorganic urea nitrogen fertilizer were sinks of CH_4 in the eastern U.S. Corn Belt.

3.2. Nitrous Oxide (N_2O) Flux. As with soil CH_4 flux data, there was a significant variation in soil N_2O fluxes within and among tillage and fertilizer treatments (Figure 3).

There was a significant seasonal variation in soil N_2O fluxes among tillage and N fertilizer treatments (Table 4). There was no significant difference in soil N_2O fluxes between CT and NT systems which had mean N_2O fluxes of 20.99 and 23.70 g N_2O ha^{-1} day^{-1}, respectively, in spring. However, the mean soil N_2O flux of 50.45 g N_2O ha^{-1} day^{-1} in grass fallow plots was about twice that for plots under CT and NT systems in spring (Table 4). Johnson et al. [31] reported high N_2O fluxes in spring but found no significant differences in annual cumulative N_2O fluxes between CT and NT systems. In Minnesota, Venterea et al. [47] found no significant differences in N_2O emissions between CT and NT systems when using urea ammonium nitrate fertilizer in corn plots.

A mean soil N_2O emission of 14.50 g N_2O ha^{-1} day^{-1} was observed in the NT system in summer, whereas plots under CT and grass fallow systems were sinks of N_2O with fluxes of −37.60 and −71.80 g N_2O ha^{-1} day^{-1}, respectively. In fall, plots under CT and NT systems had soil N_2O fluxes of −19.20 and −106.30 g N_2O ha^{-1} day^{-1}, respectively, whereas the grass fallow plots had a mean soil N_2O emission of 122.0 g N_2O ha^{-1} day^{-1} (Table 4). In winter, plots under CT system had a mean soil N_2O emission of 45.05 g N_2O ha^{-1} day^{-1}, whereas NT and grass fallow plots were sinks of N_2O with flux rates of −19.27 and −14.95 g N_2O ha^{-1} day^{-1}, respectively.

Seasonal variation in soil N_2O fluxes has been reported by other researchers. Our results are in agreement with the findings of Jacinthe and Dick [20] who reported that seasonal N_2O N losses from chisel-tilled plots were generally significantly higher than those from NT plots. Jacinthe and Dick [20] also reported average daily N_2O emissions ranging from 0.1 to 326 g N_2O ha^{-1} day^{-1} in a corn/soybean (*Glycine max* L.)/wheat (*Triticum aestivum* L.)/hairy vetch (*Vicia villosa* Roth) rotation study, with seasonal N_2O N losses which were highest in continuous corn plots and lowest in soybean plots. Hernandez-Ramirez et al. [46] found significant seasonal variation in N_2O emissions in an eastern Corn Belt soil, while Almaraz et al. [48] found higher N_2O fluxes during the spring which were associated with precipitation events in a study in Quebec. In our study, we did not find significant correlations between soil N_2O fluxes and soil temperature and moisture conditions during measurements.

Mean N_2O fluxes over seasons during the study period show that CT and grass fallow plots had net emissions of 2.31 and 21.42 g N_2O ha^{-1} day^{-1}, respectively, whereas plots under NT system were net sinks of atmospheric N_2O with a mean flux rate of −21.84 g N_2O ha^{-1} day^{-1} (Table 4). Our study on the Decatur silt loam soil in north Alabama therefore shows that while CT system can be an insignificant source of soil CH_4, it can be a significant source of soil N_2O which is a more potent greenhouse gas than CH_4. On the other hand, an NT cropping system can reduce the agricultural contribution of both CH_4 and N_2O, since plots under NT were sinks of CH_4 and N_2O. In the Midwest, Kessavalou et al. [49] reported that an NT fallow system exhibited the least threat to cause the deterioration of atmospheric quality due to greater CH_4 uptake and decreased N_2O emissions.

Contrary to our findings, Almaraz et al. [48] suggested that changing from CT to NT system under the heavy soil conditions of Quebec may increase greenhouse gas contributions mainly as result of the increase in N_2O emission. However, the above authors indicated that this negative effect of NT could be reduced by avoiding fertilizer application during high precipitation periods, thus indicating that the results were mainly due to the effect of soil moisture conditions at fertilizer application time. As reported by Mosier et al. [43], N_2O emissions can be variable depending on soil moisture conditions, N mineralization, and plant community dynamics, which can explain the variability in reported findings on effect of tillage systems on N_2O emissions.

Plots which received AN-SI, PL-SA, and PL-BA had N_2O emissions of 240.25, 20.51, and 19.20 g N_2O ha^{-1} day^{-1}, respectively, in spring, whereas plots which received AN-SA, PL-SI, and the control plots were N_2O sinks (Table 4). There was no significant difference in soil N_2O fluxes between plots which received PL-SA and PL-BA in spring. In summer, plots which received AN-SA, AN-SI, and PL-BA and the control plots were net sinks of N_2O, whereas plots which received PL-SI and PL-SA had N_2O emissions of 86.43 and 117.09 g N_2O ha^{-1} day^{-1}, respectively. In fall, plots which received AN-SA, PL-SA, and PL-BA and the control plots were sinks of N_2O, whereas plots which received AN-SI and PL-SI had N_2O emissions of 211.48 and 168.32 g N_2O ha^{-1} day^{-1}, respectively (Table 4). In winter, plots which received PL-SA, 150 kg N ha^{-1} PL-BA, and the control plots were sinks of N_2O, whereas plots which received AN-SA, AN-SI, and PL-SI had large N_2O emissions of 735.00, 89.40, and 186.98 g N_2O ha^{-1} day^{-1}, respectively (Table 4).

Averaged over tillage and N fertilizer treatments, N_2O oxide emissions of 32.14 and 21.37 g N_2O ha^{-1} day^{-1} were observed in spring and winter, whereas in summer and fall, the plots were sinks of N_2O with fluxes of −30.23 and −7.23 g N_2O ha^{-1} day^{-1}, respectively (Table 4). Mean soil N_2O fluxes in each N fertilizer treatment averaged over seasons during the study period show that the 0N control and the PL-BA treatments were huge net sinks of N_2O with a flux of about −141.63 g N_2O ha^{-1} day^{-1}, whereas all plots under inorganic AN fertilizer treatments, the PL-SA, and the PL-SI treatments had net N_2O emissions (Table 4). The three highest cumulative soil N_2O emissions were observed in plots which received AN-SA (173.68 g N_2O ha^{-1} day^{-1}), PL-SI (100.55 g N_2O ha^{-1} day^{-1}) and AN-SI (55.19 g N_2O ha^{-1} day^{-1}).

The amounts of N_2O emitted from soils depend on complex interactions between soil properties (such as soil aeration status, temperature and carbon availability, soil texture), type and management of N fertilizer preceding crop, residue management, and other agricultural practices as well as prevailing climatic conditions [50, 51]. Despite being a source

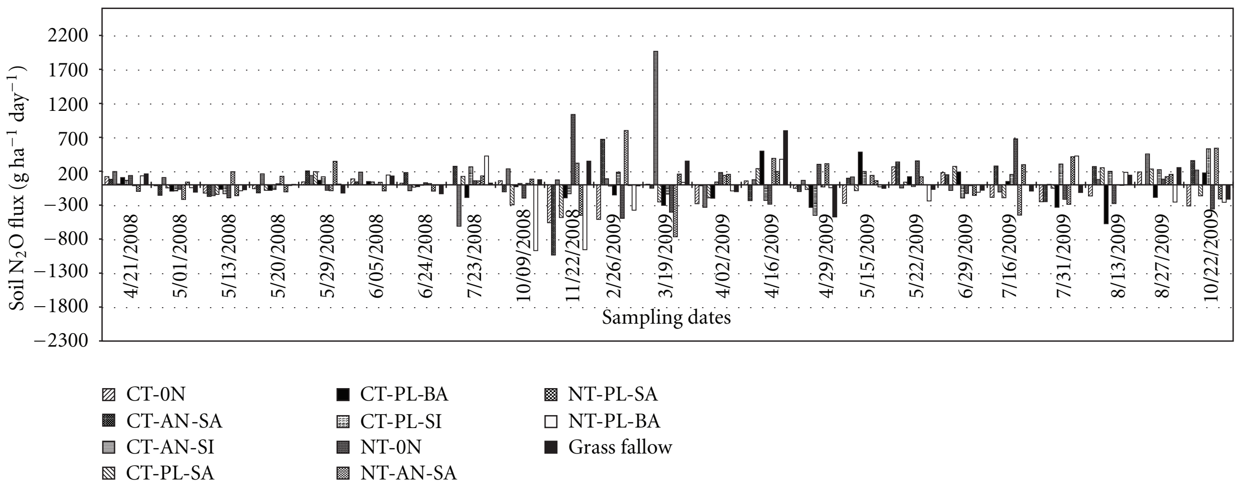

FIGURE 3: Soil N_2O fluxes in corn plots receiving ammonium nitrate (AN) and poultry litter (PL) N fertilizers using surface (SA), soil incorporation (SI), and subsurface band (BA) application methods in conventional tillage (CT) and no-tillage (NT), systems, Hazel Green, Alabama.

TABLE 4: Soil N_2O fluxes as influenced by tillage systems and ammonium nitrate (AN) and poultry litter (PL) N fertilizers using surface (SA), soil incorporation (SI), and subsurface band (BA) application methods in spring, summer, fall, and winter seasons, Hazel Green, Alabama.

	Spring	Summer	Fall	Winter	Mean flux
			N_2O flux (g ha^{-1} day^{-1})		
Tillage systems					
Conventional tillage	20.99a[†]	−37.60b	−19.20b	45.05c	2.31
No-till	23.70a	14.50c	−106.30a	−19.27a	−21.84
Grass fallow	50.45b	−71.80a	122.00c	−14.95b	21.42
Fertilizer treatments					
0N control	−14.15b	−8.20d	−42.34c	−501.82a	−141.63
150 kg N ha^{-1} AN-SA	−0.22c	−15.19c	−25.14d	735.28f	173.68
150 kg N ha^{-1} AN-SI	240.25e	−320.38a	211.48f	89.40d	55.19
150 kg N ha^{-1} PL-SA	20.51d	86.43e	−88.04b	−14.07c	1.21
150 kg N ha^{-1} PL-SI	−70.20a	117.09f	168.32e	186.98e	100.55
150 kg N ha^{-1} PL-BA	19.20d	−44.69b	−305.04a	−260.97b	−147.94
Seasonal means	32.14	−30.23	−7.32	21.37	

[†] Means for tillage system or N fertilizer treatment within a season followed by the same letter are not significantly different at the $P \leq 0.05$ level.

of N_2O, the soil can also remove atmospheric N_2O under conditions favorable for N_2O reduction [52, 53]. Although this is probably only a minor sink on the global scale, the elimination of N_2O in the stratosphere is so slow that even a small soil sink can contribute significantly to reduce the atmospheric residence time of N_2O [51]. The current lack of literature documenting the behavior of soils acting as sinks of N_2O is most probably due to the fact that current studies in agricultural ecosystems are not only designed to but are also more focused on measuring soil, emissions of N_2O as opposed to soil N_2O uptake, since emissions are linked to climate change.

Our results clearly show that plots which received AN fertilizer were net emitters of soil N_2O irrespective of application method. Nitrous oxide is produced in soils as an intermediate during nitrification and denitrification processes [11, 54, 55]. Therefore N application, especially, inorganic fertilizer sources in which NO_3-N is readily available in the soil can generally result in more N_2O emissions compared to control plots, as indicated by high N_2O fluxes in plots which received 150 kg N ha^{-1} AN using surface or soil incorporation application methods. Similarly, on a clay soil in Canada, Chantigny et al. [56] found significantly greater cumulative soil N_2O-N emissions with mineral fertilizer treated soil compared to liquid swine manure, which is also an indication that soil N_2O production is primarily driven by availability of NO_3-N in the soil. Although N fertilizer in the form of PL using soil incorporation (150 kg N ha^{-1} PL-SI) application method had a negative mean soil CH_4 flux of −14.97 g CH_4 ha^{-1} day^{-1}, the high mean N_2O emissions of 100.55 g N_2O ha^{-1} day^{-1} show that this treatment is not sustainable. On the other hand, subsurface band application of PL resulted in net negative CH_4 and N_2O fluxes of −8.36 g CH_4 ha^{-1} day^{-1} and −147.94 g N_2O ha^{-1} day^{-1}, indicating that subsurface band application of PL can be a sustainable poultry waste management practice for reducing agricultural greenhouse gas contributions to the atmosphere.

Interestingly, plots with surface application of PL (PL-SA treatment), which is the standard practice for PL application in NT system, were net sinks of CH_4 and small emitters of soil N_2O over the study period. This suggests that given the other benefits of PL such as supplying nutrients for plant growth, improving soil quality through direct addition of C and N to the soil [1], surface application of PL in cover crop residues could be more environmentally sound compared to soil incorporation. However, the extra benefits of significantly higher CH_4 and N_2O uptake observed with subsurface band application of PL far out-weigh the benefits of surface application of PL, which implies that subsurface band application could be the best animal waste management strategy for reducing agricultural greenhouse gas emissions.

Under anaerobic or high soil moisture conditions, NO_3-N from N fertilization with an organic fertilizer such as PL can be associated with an increased rate of N_2O production from denitrification thereby resulting in soil emissions of N_2O. In order for N_2O production from denitrification to be significant, O_2 consumption by soil microbes must exceed O_2 diffusion into the soil, thereby creating microsites of anaerobiosis [57]. Organic matter in animal manure can increase percent water-filled pore spaces in mineral soils which can reduce soil aeration and decrease O_2 diffusion into the soil thereby leading to anaerobic conditions. Linn and Doran [29] found that higher percentage of water-filled pores in NT soils was reflected in higher N_2O production and that N fertilization increased N_2O production irrespective of tillage system.

In this study, we found that N_2O emissions directly above the PL band were generally higher compared to those where the PL was mixed with the soil under SA or SI methods, that is, where there was no PL band. This is due to the fact that subsurface band application of PL application concentrates the PL in a narrow strip (4.4 cm wide) at a depth of about 7 cm under the soil surface at a 97 cm band spacing. This increases the amount of C and N substrates in a confined area, hence an increase in the amount of NO_3 from mineralization of the PL. Poultry litter also causes the soil to hold more water and retain the moisture longer and thereby increasing both the rate of mineralization and the percent water-filled pore space, which promotes anaerobic conditions which promote N_2O production from denitrification. However, despite the higher N_2O emissions directly above the PL band, the "effective" per plot N_2O fluxes, taking into account the fact that the PL bands cover only narrow strips of soil in the plots, were significantly lower than N_2O fluxes in plots which received PL using the soil incorporation method.

Engel et al. [58] reported that placement of urea fertilizer in nests or bands resulted in delayed but prolonged soil N_2O emissions compared to surface broadcast applications on an Amsterdam silt loam soil in Montana. On a Waukegan silt loam soil in Minnesota, Venterea et al. [47] reported that surface broadcasting urea in an NT system resulted in higher N_2O emissions compared to CT, whereas soil injection of anhydrous ammonia resulted in higher N_2O emissions under CT system. Alluvione et al. [59] found that the slow mineralization of composted manure in spring reduced N_2O emissions compared to urea fertilizer and that after combining N_2O and CO_2 fluxes, compost reduced the CO_2 equivalent emissions by 49% compared to urea application. Rochette et al. [60] found double N_2O emissions in NT plots compared to tilled plots in heavy clay soils but found no difference between tillage systems on loam soils.

Some studies have reported higher N_2O emissions in NT compared to CT cropping systems [61–63], while some studies have found lower emissions in NT soils or no difference between tillage systems [64, 65]. Sistani et al. [66] reported that application of effluent liquid manure below the soil surface by injection resulted in elevated CH_4 emissions compared to the control treatment, while Drury et al. [67] found that reduced tillage and shallow N placement depth reduced N_2O emissions on a clay loam soils in eastern Canada. The variations in research findings discussed above suggest that soil and fertilizer management practices can result in different responses in soil emissions of greenhouse gases depending on soil type, fertilizer type, and soil environmental conditions. Therefore, site-specific tillage and fertilizer management practices for different locations need to be developed and evaluated in order to achieve the goal of reducing agricultural greenhouse gas emissions at local, regional, and global scales.

4. Conclusion

There were significant seasonal variations in soil CH_4 and N_2O emissions among tillage and N fertilizer treatments in corn plots in north Alabama. Mean soil CH_4 fluxes over seasons during the two-year study period in CT and NT systems were not significantly different. Mean soil CH_4 and N_2O fluxes averaged over tillage and N fertilizer treatments and seasons showed that the plots were net sinks of CH_4 and net emitters of N_2O over the two-year study period. Averaged over seasons, plots under NT were net sinks of soil N_2O, while CT plots and grass fallow plots were net emitters of soil N_2O. Plots which received AN fertilizer were net emitters of soil CH_4 and N_2O, during the study period, irrespective of application method, whereas plots which received PL fertilizer were net sinks of CH_4, irrespective of application method. Plots which received PL using surface or soil incorporation application methods were net emitters of N_2O, while plots which received PL using subsurface band application method were net sinks of N_2O. Our study indicates that using subsurface band application of PL was the most promising environmentally sustainable poultry waste application method for reducing CH_4 and N_2O emissions from agricultural soil in NT and CT corn production systems on the Decatur soil in north Alabama.

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