

AGROECOSYSTEM MANAGEMENT EFFECTS ON CARBON AND NITROGEN
CYCLING ACROSS A COASTAL PLAIN CATENA

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AGROECOSYSTEM MANAGEMENT EFFECTS ON CARBON AND NITROGEN
CYCLING ACROSS A COASTAL PLAIN CATENA

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DISSERTATION ABSTRACT

AGROECOSYSTEM MANAGEMENT EFFECTS ON CARBON AND NITROGEN
CYCLING ACROSS A COASTAL PLAIN CATENA

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Agricultural activities contribute an annual increase in radiative forcing of about 20%. In southeastern US, use of cover crops in conservation tillage (CsT) has increased in recent years. However, data on soil C and N dynamics and subsequent trace gas emissions at the landscape scale are lacking. Objectives of this study were to evaluate effects of landscape and soil management on 1) methane (CH₄), nitrous oxide (N₂O) and carbon dioxide (CO₂) fluxes, 2) soil carbon (C) and nitrogen (N) mineralization and 3) cover crop decomposition and mineralization.

Gas fluxes, C and N mineralization, and cover crop decomposition were determined on a 9-ha field at the E.V. Smith Research Center near Shorter, in AL. This experiment consists of six replications of agroecosystem management [(corn (*Zea mays* L.)-cotton (*Gossypium hirsutum* L). rotation] that traverse the landscape. Soil managements included CsT, conventional tillage (CT), conservation tillage with dairy

manure (CsTM), and conventional tillage with dairy manure (CTM) treatments. The soil management treatments were within summit, sideslope and the drainageway landscape positions.

The drainageway landscape position emitted 46, 251, 59, and 185 mg CH₄-C ha⁻¹ h⁻¹ from CT, CTM, CsT and CsTM treatments, respectively. The summit position was a CH₄ consumer with CT and CsT treatments. Significant soil management treatment differences in N₂O-N flux were observed only within the drainageway landscape position. Averaged across seasons, CT and CsT emitted similar N₂O-N in the drainageway. Within the drainageway, dairy manure decreased N₂O-N emission on CT treatments. Carbon dioxide emission in winter 2005 from CsT treatments (averaged across landscape positions) was 1304 g ha⁻¹ h⁻¹ CO₂-C compared to 227 g ha⁻¹ h⁻¹ CO₂-C from CT treatments.

CsT and CsTM treatments increased soil organic C and total soil N after six years. This resulted in higher C and N mineralization on soils from CsT and CsTM treatments, with no differences between landscape positions.

Potential C mineralization was similar for crimson clover, spring forage rape and white lupin amended soil while black oat amended soil immobilized N. Buried cover crops decomposed and mineralized faster than surface applied materials, with no differences in cover crop decomposition and mineralization *k* across landscape positions.

Overall, landscape variability had minimal effect on C and N dynamics and cover crop decomposition compared to soil management effects. Conservation tillage, dairy manure applications, and cover crops showed potential to sequester soil organic C and increase total soil N in these systems.

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I. INTRODUCTION

In order to meet the world's increasing food and energy demands, there is a need to intensify agricultural production. This implies opening more agricultural land or increasing production on current sites. Either of these alternatives has implications for greenhouse gas (GHG) emissions. Cultivating forest land for agriculture leads to increased C fluxes through increased mechanization and fertilizer use (West and Marland, 2003). Common soil management practices used in agricultural food production include tillage, use of cover crops and application of fertilizers. These practices influence GHG emissions that are linked to climate change.

Greenhouse gases include methane (CH_4), nitrous oxide (N_2O) and carbon dioxide (CO_2). Methane is an end product of anaerobic bacterial decomposition of plant and animal litter in environments where nitrate (NO_3) and sulphate (SO_4) concentrations are low (Le Mer and Roger, 2001). The main sources include wetlands, landfills and municipal solid waste landfills. In the troposphere, absorption of CH_4 occurs through reactions with OH radicals that break it into CH_3 and water vapor (Hütsch, 2001). In the soil, CH_4 is oxidized to CO_2 by methanotrophic bacteria. This is beneficial to the environment since CO_2 has 32 times lower radiative potential than CH_4 (Hütsch, 2001).

Nitrous oxide is a trace gas that contributes to atmospheric greenhouse gas concentrations (Dickinson and Cicerone, 1986) and arises from denitrification and

nitrification in agroecosystems (Davidson, 1992). It is also involved in destruction of stratospheric ozone (Cicerone, 1987). There is no significant mechanism for consumption of N₂O in agricultural systems, and as a result, mitigation focuses on emission reduction.

Carbon dioxide arises from root respiration and microbial respiration in soil during breakdown of soil organic matter (SOM). Reduction in CO₂ emission from soil may be achieved through use of reduced tillage systems.

The Greenhouse Effect

Methane, N₂O, CO₂, and other greenhouse gases occur naturally in the atmosphere. Short wavelength radiation from the sun passes through these gases and reaches the earth's surface where it is converted to heat (Jacob, 1999). However, these greenhouse gases do not allow most of the reflected infrared radiation (long wave) from the earth's surface to pass through the earth's atmosphere back to space. Instead, the gases absorb most of the radiation. Gas molecules absorb radiation of a given wavelength only if the energy can be used to increase the internal energy of the gas molecule (Jacob, 1999). Carbon dioxide, CH₄, and N₂O gas molecules can acquire charge symmetry by stretching or flexing resulting in changes of dipole moments of the molecules. The change in the dipole moment enables the gas molecules to absorb radiation in the near infrared (wavelength = 0.7-20 μm). Most terrestrial radiation is emitted at 5-50 μm, and gases that absorb radiation at this range are known as greenhouse gases. Some of the radiation trapped by the greenhouse gases is radiated back to the earth's surface resulting in climate change (IPCC, 1990). This greenhouse effect is

necessary to some extent, as it allows the earth to warm and sustain life on earth (Hütsch, 2001). However, increasing greenhouse gas concentrations result in an increased greenhouse effect that eventually results in climate change. Studies show that levels of greenhouse gases have been increasing since large-scale industrialization began in the 1750s (IPPC, 2001).

Tillage Effect on Greenhouse Gas Emissions

Carbon Dioxide

Tillage operations increase soil aeration and exposure of SOM to microbial populations (Paustian et al., 1997). This increases CO₂ release through microbial respiration and promotes a reduction in SOM. In a review of CO₂ emissions of Australian agricultural land, So et al. (2001) showed that conventional tillage contributed to greater SOC loss and CO₂ emissions compared to no-till. They estimated that tillage operations in Australia produce an average of 20 g m⁻² CO₂ after a single tillage operation of moist soil. Reicosky (1991) found that intensive tillage using moldboard plowing in Minnesota resulted in 81 g CO₂ m⁻² released five hours after tillage operation, compared to no-till that produced only 6 g CO₂ m⁻² over the same time period. He also compared tillage implements (moldboard plow vs. disk harrow adjusted to vary residue cover remaining after tillage) that leave >30% residue cover on the surface following operation and found that use of these implements resulted in lower CO₂ flux compared to inversion tillage. However, use of these implements resulted in more CO₂ emission compared to strict no-till.

Six et al. (2004) reviewed published data that compared no-till and conventional tillage soil C sequestration (254 data points). They suggested net C sequestration occurred early on adoption of no-till in humid climates, whereas net C sequestration was only achieved after 20 years of no-till adoption in drier climates.

Nitrous Oxide

The review by Six et al. (2004) used published data to compare the effect of no-till and conventional tillage on N₂O emissions. They found N₂O emissions increased following no-till adoption for 10 years in both humid and dry climates. Increased N₂O emission under no-till was attributed to higher soil water content that favors denitrification. After 20 years of no-till, N₂O emissions decreased in humid climates but remained similar between conventional and no-till systems in dry climates.

Methane

Six et al. (2004) found five studies that compared CH₄ fluxes between no-till and conventional tillage. In these studies, no-till systems increased CH₄ uptake by 0.6 kg ha⁻¹ yr⁻¹. They attributed this to greater pore continuity and presence of ecological niches for methanotrophic bacteria that develop in no-till systems relative to conventional tillage systems. Keller et al. (1990) found that cultivated agricultural soils in central Panama oxidized less CH₄ than non-cultivated forest soils. Rates of CH₄ oxidation in agricultural soils were one-fourth those of undisturbed forest soil. Lower rates of CH₄ oxidation in cultivated soils may be due to disturbance of the ecological niche for methanotrophic bacteria (Willison et al., 1995).

Manure Effects on Greenhouse Gas Emissions

Nitrous Oxide

Use of animal manure as a soil amendment influences N_2O fluxes. Clayton et al. (1997) found that application of cattle slurry supplemented with ammonium-nitrate near Edinburgh, Scotland, resulted in higher N_2O emission compared to ammonium-nitrate application. They associated this with the contribution of organic C and water in the slurry, factors that favor N_2O emission through denitrification. In the same study, application of slurry alone resulted in N_2O four times higher than in non-fertilized plots. Farrel et al. (2003) compared the effect of manure application and urea N sources on N_2O emissions in the Canadian prairies over a two year period. They found that across years, both N sources increased N_2O emissions above background levels. Ginting et al. (2003) found no significant effect of manure application on N_2O emissions four years after application of manure and compost in Nebraska (US) soils.

Methane

Long-term application of farmyard manure has been found to inhibit CH_4 oxidation (Hütsch, 2001). This effect has been attributed to the fraction of N from manure that is easily ammonified. Ammonium is detrimental to methanotrophic bacteria growth and reproduction. According to Hütsch (2001), this N fraction is small, and increase in microbial biomass that result from long-term application of farmyard manure result in increases of methanotrophic bacteria. This would counteract the inhibiting effect of N on CH_4 oxidation in manure. Willison et al. (1995) found that mowed grass had 80% higher CH_4 oxidation than grazed land on a long-term experiment in

Rothamsted, UK. They attributed lower CH₄ oxidation on the grazed site to N supplied via urine and feces by grazing sheep (*Ovis aries* L.). This added N decreased CH₄ oxidation.

Carbon dioxide

Manure is applied to agricultural soils as slurry or as solid material. The two forms have different effects on CO₂ emissions due to differences in nutrient and moisture content. Higher rates of CO₂ emissions were observed after application of liquid manure to a loamy soil than when solid manure was applied (Rochette et al., 2006). Manure-induced surface CO₂ emissions are large following manure application, and decrease with time after application (Rochette et al., 2006 and Gregorich et al., 1998).

Fall and spring injection of pig slurry on a Canadian soil resulted in short lived increase in microbial biomass lasting 25 days, and a similarly short lived CO₂ flush in both seasons (Rochette et al., 2004a). Higher CO₂ flush occurred after spring application than after the fall application due to slurry carbonate dissociation on contact with the acid soil. Lower CO₂ flush in fall compared to spring was due to higher soil moisture and lower temperatures in fall than in spring, that may have resulted in higher CO₂ solubilization in soil water. In Central Canada, CO₂ emission from manure applied in spring on a corn (*Zea mays* L) field increased with increasing rate of manure application (Gregorich et al., 1998). However, relative increase in CO₂ emission on doubling manure rate from 56 to 112 Mg ha⁻¹ was small, suggesting that application of high manure rates resulted in higher soil C storage and lower CO₂ emission rate per unit manure applied.

Gregorich et al. (1998) attributed this to possible O₂ limitation at the high manure application rate.

Inorganic Fertilizer Effects on Greenhouse Gas Emissions

Nitrous Oxide

Nitrous oxide emission from soil occurs through the processes of nitrification and denitrification (Davidson, 1992). Nitrification requires mineral N as substrate (Mosier et al., 1998). Consequently, addition of N fertilizers to soil enhances N₂O emission (Hall and Matson, 2003; Karen and Smith, 2003; Clayton et al., 1997). In a study in Great Britain, Karen and Smith (2003) observed higher gas emissions at the onset of fertilizer application that decreased over time. They also found consistently higher gas emissions on grassland than on cropped land. Pomes et al. (1998) estimated N losses in a Missouri (US) soil to range between 2.4-4.3% of the N applied as fertilizer. Clayton et al. (1997) found consistently higher N₂O emissions on fertilized plots than on on-fertilized plots on poorly drained soils near Edinburgh, Scotland.

Conversely, Kim and Kim (2002) found no ammonium-nitrate fertilizer effect on N₂O fluxes on a Korean soil. However, they found an increase in NO emission upon addition of fertilizer. They also found higher N₂O fluxes under upland rice (*Oryza sativa*) than under paddy rice (. They attributed lower N₂O fluxes under paddy rice to possible reduction of N₂O to N₂ under flooded conditions. Similarly, other studies have shown that soil water saturation influences N₂O emission (Karen and Smith, 2003; Pomes et al., 1998; Ball et al. 2002; Clayton et al., 1997; Farrell et al. 2003). Pomes et al. (1998) observed that most N₂O fluxes occurred at a soil gravimetric water content of 77-89% in

some Missouri (US) soils. Gas emissions decreased when soils were near saturation. Clayton et al. (1997) found that N₂O emissions decreased at saturated porosity greater than 90%. Schmid et al. (2001), in a study conducted in Switzerland, found that after a heavy precipitation N₂O emission decreased temporarily and peaked again when the soil dried. At soil water content near saturation, N₂O is reduced to N₂ before it volatilizes (Clayton et al., 1997). Ball et al. (2002) found that rainfall increased N₂O fluxes after fertilizer additions on Scotland soils. Fluxes increased after rainfall and remained high for 20 days during which mineralization and denitrification of N were likely to have been rapid. Farrell et al. (2003) found that rainfall distribution influenced N₂O fluxes. They obtained higher cumulative N₂O emission when rainfall was evenly distributed across the season.

Hall and Matson (2003) found negligible N₂O emissions after first-time addition of N fertilizer in N limited tropical forest ecosystems. However, long-term N fertilizer additions resulted in increased N₂O fluxes. Evidently, addition of N fertilizers when N was not limiting primary production resulted in higher N₂O emissions. Their laboratory studies further showed higher losses occurred with ammonium N fertilizer than with nitrate N fertilizer. Clayton et al. (1997) compared effects of N fertilizers on N₂O fluxes, and found that urea application gave higher summer N₂O emissions than calcium nitrate and ammonium nitrate.

A study in Argentina by Rozas et al. (2001) established that application of N fertilizer to irrigated corn (*Zea mays* L.) at the six-leaf stage resulted in less N₂O emission than application at planting. Application of 70 kg N ha⁻¹ and 210 kg N ha⁻¹ at the six-leaf stage resulted in similar accumulated N₂O–N losses (2.0 and 2.1 kg N ha⁻¹, respectively),

while when the same N rates were applied at planting, accumulated N₂O-N loss of 7.6 and 9.8 kg N ha⁻¹, respectively, were obtained. The lower N loss at the six-leaf stage was probably due to N uptake by the corn crop. However, the fraction of N lost was lower for the higher N application rate than for the lower rate for both applications. Rochette et al. (2004b) studied the effect of N rates on N₂O emission in poorly drained Canadian soils using anhydrous ammonia in corn production. They found no clear short-term effect of excess N addition on N₂O emission. They suggested that in their study, N₂O dynamics were limited by factors other than soil N availability.

Methane

In a study in Rothamsted, UK, Willison et al. (1995) found that application of ammonium sulphate resulted in complete inhibition of CH₄ oxidation. However, application of NO₃ had no effect on oxidation relative to a control (no N application). Chan and Parkin (2001) found that field application of urea-ammonium-nitrate did not inhibit CH₄ production. According to Seghers et al. (2003), NO₃ decreases low affinity CH₄ oxidation, while ammonium decreases high affinity CH₄ oxidation. High affinity CH₄ oxidation occurs at CH₄ concentrations close to that of the atmosphere, while low affinity CH₄ oxidation occurs at higher CH₄ concentrations (Le Mer and Roger, 2001). Also, repeated addition of different fertilizer treatments can change the community structure of methanotrophs (Seghers et al., 2003).

Carbon dioxide

Inorganic fertilizers affect soil CO₂ emission through their influence on soil microorganisms and plant root respiration. In field studies in South Dakota and eastern Montana (US), Sainju et al. (2008) reported increased CO₂ emission from barley (*Hordeum vulgare* L.), rye (*Secale cereale* L.) and winter pea (*Lathyrus odoratus* L.) fields treated with urea fertilizer. They attributed this to increased root and microbial respiration due to increased crop growth.

Following four years of wheat straw (*Tritium aestivum* L.) application under no-till and no crops grown in Iowa, urea fertilizer had no effects on CO₂ emission (Jacinthe et al., 2002). According to findings of Lee et al. (2007), fertilization with ammonium nitrate had no effect on CO₂ flux and soil microbial biomass on a switchgrass (*Panicum virgatum* L.) field in South Dakota (US). Similarly, Al-Kaisi et al. (2008) found no effect of broadcast ammonium nitrate fertilizer on CO₂ emission on a corn/soybean (*Glycine max* L.) rotation in four sites in Iowa. Increase in soil temperature and moisture content led to increased CO₂ emission rates. However, laboratory incubation of soil from the study sites indicated that cumulative CO₂ emission decreased with increasing N application with highest emission rates observed on no-N plots.

Crop Residue Effect on Greenhouse Gas Emissions

Nitrous Oxide

Mineralization of organic residues influences N dynamics. Huang et al. (2004) found an increase in N₂O emissions with addition of crop residues in a laboratory incubation study. They used five residues of different qualities and found that N₂O

emission was negatively correlated with C:N ratio. They suggested that residues with low C:N ratio decomposed faster, providing a greater opportunity for release of dissolved organic C, a resource for microbial growth. Addition of urea to crop residues increased N₂O emission for all residues except sugarcane (*Saccharum officinarum* L.) residue, where a decrease in N₂O emission was observed. This was thought to be due to microbial immobilization of added N due to the large C:N of sugarcane (*Saccharum officinarum* L.) residue.

Methane

Crop residues can influence CH₄ emissions through their influence on soil moisture content as well as soil NH₄-N concentration. Crop residues with a high C:N ratio immobilized soil N and had no effect on CH₄ oxidation, while residues with low C:N ratio enhanced N mineralization and strongly inhibited CH₄ oxidation (Hütsch, 1998). This inhibition was caused by gradual accumulation of NH₄-N during crop residue decomposition. Similar results were reported by Boeckx and Van Clement (1996) in a laboratory study using wheat and corn residues (high C:N ratio), and potato (*Solanum tuberosum* L.) and sugar beet (*Beta vulgaris* L.) residues (low C:N ratio).

Carbon dioxide

Crop residues can influence CO₂ emission by altering soil properties and by acting as a physical barrier that reduces diffusion of CO₂ from the soil to the atmosphere. Leaving crop residues on the soil surface was found to decrease CO₂ emission in both conservation and conventional tillage systems in Iowa (US) (Al-Kaisi and Yin, 2005).

This could have been due to crop residue acting as a barrier for CO₂ emission from soil to the atmosphere, and due to lower decomposition rates as a result of reduced residue-soil contact.

In contrast, four years of wheat residue amendments increased CO₂ emission in Ohio (US) (Jacinthe et al., 2002). Higher CO₂ emission fluxes were observed in late winter and summer, and were related to changes in soil temperature. Notable was temporal influence of wheat mulch on seasonal variation in CO₂ emission with temperature changes. Mulched plots showed a delayed increase in CO₂ emission as air temperature and consequently soil temperature increased in late winter. In a greenhouse study, rice (*Oryza sativa* L.) residues (under paddy conditions) increased CO₂ emission (Lou et al., 2007). Emissions were correlated with microbial biomass C and soluble C, and were high at initial stages of residue application (< 25 days), but gradually decreased with time. Rice straw increased CO₂ emission above levels observed with rice root amendments due to higher cellulose and lower lignin concentration in straw compared to roots.

Landscape Effect on Greenhouse Gases

Nitrous Oxide

Landscape position may influence greenhouse gas emissions. This is due to differences in soil properties, soil moisture dynamics and nutrient availabilities found on different landscape positions. Sehy et al. (2003) compared N₂O fluxes on corn fields located on foot slope and shoulder positions managed with precision farming in Munich, Germany. Nitrogen fertilizer was applied according to crop requirement with the foot

slope areas receiving 175 kg N ha^{-1} while the shoulder positions received 125 kg N ha^{-1} . They found higher N_2O emissions on foot slope positions compared to shoulder positions. They attributed the difference to higher water filled porosity ($>60\%$) at the foot slope position resulting from lateral downslope water movement. They ruled out soil textural effects as the two areas had similar textures. Farrell et al. (2003) studied the effect of soil management and landscape positions on N_2O emissions on some Canadian soils. They used swine manure and urea as N sources and compared tillage practices and fertilizer rates on different slope positions. They found that effects of management practices and fertilizer rates on N_2O emission were influenced by slope position. Nitrous oxide emissions were greatest on the low catchment areas that consisted mainly of footslopes and lower lying level positions. They further observed that in terms of N_2O production, the sideslopes performed like shoulders and summits during dry weather, and like foot slopes during wet periods. This effect was attributed to water redistribution within the landscape.

Methane

Chan and Parkin (2001) used a closed chamber method to compare CH_4 emissions in cultivated and natural ecosystems in central Iowa (US). Generally, CH_4 production under cultivated land was higher than under natural vegetation. On a no-till agricultural site, chambers were placed 10-m apart along a transect traversing low and high areas of the field. They observed that chambers located at lower elevations tended to exhibit positive CH_4 fluxes, while those at the higher elevations showed negative fluxes. Thus,

higher elevation locations were net consumers (through CH₄ oxidation) while lower elevations were net producers of CH₄.

Landscape and Soil Variability Effect on Soil C and N Mineralization

Neil et al. (1997) studied N mineralization across a 700 km transect over a geographical range in the Brazilian Amazon. They sampled soils (Oxisols and Utisols) in a chronosequence of forest and a young pasture and measured for total C, NH₄-N, NO₃-N and N mineralization in a laboratory incubation study. They found that under native forest conditions, net N mineralization and nitrification were higher in soils high in clay. Soils high in clay content were also higher in organic matter, which stimulated N mineralization and nitrification. Pastures established from forest clearing showed lower net N mineralization and nitrification compared to forest. This implies that soils under pastures established after clearing forests have less N₂O emissions compared to soils under the original forest. Frank and Groffman (1998) studied *in situ* N mineralization by burying soils in polythene bags for one year. They found that sites at the bottom of the slope had higher moisture, N and C, than summits. Lower landscape positions showed higher cumulative net N mineralization and cumulative C respiration relative to summits. Morris and Boerner (1998) studied the effect of topography on N mineralization and nitrification in a watershed scale. They stratified the watershed using a GIS-based integrated moisture index, and developed three moisture classes per watershed (xeric, intermediate and mesic) in a hardwood forest ecosystem. Soil was sampled from each of the moisture classes and analyzed for nitrification and potential NO₃ mineralization. Results showed that nitrification, potential NO₃ mineralization, organic C, NH₄ and pH

were significantly lower for the xeric class than for intermediate and mesic classes. This was probably due to less soil moisture at the xeric moisture class. On agricultural land, Wood et al. (1990) found lower C turnover and relative N mineralization rates on foot slope positions compared to summit and back slope positions. They speculated this was caused by greater accumulation of recalcitrant C and N organic compounds at the lower landscape positions.

Cover Crop Decomposition

Cover crops are grown during the winter season to protect soil from erosion, improve soil properties and retain soil nutrients. As they decompose, they supply nutrients to a subsequent crop. Decomposition is governed by material quality, environmental factors, and soil organisms (Swift et al., 1979). A study by Ruffo and Bollero (2003a) in Illinois soils showed that only 5% of initial mass of rye crop remained on the ground at the end of a subsequent corn growing season, while hairy vetch (*Vicia villosa* Roth) decomposed completely. This was under no-till management on land that had been in a corn-soybean rotation for five years. A litter bag decomposition study in California using cover crops showed that less than 10% of the buried material remained at the end of 16 weeks (Mitchell, 2002). The cover crops evaluated were hairy vetch (*Vicia villosa* Roth.), phacelia (*Phacelia tanacetifolia* Benth.) and barley (*Hordeum vulgare* L.), that were buried 20-cm below the soil surface with sprinkler irrigation twice a week. They demonstrated that decomposition of cover crops was rapid, and almost complete decomposition occurred within a single summer season. In north-central New Mexico, Cueto-Wong et al. (2001) used ^{15}N to determine N contribution of hairy vetch and alfalfa (*Medicago sativa* L.) to irrigated sorghum (*sorghum bicolor* [L.] Moench).

They found that only about 16 % of ^{15}N was recovered by the sorghum tops in two harvests, while 53% remained in the top 0-0.6 m soil by the end of the sorghum growing season. The 30% of the applied N that could not be accounted for was probably leached beyond the root zone or was lost to the atmosphere in gaseous form. Also, most ^{15}N recovery (80%) by the sorghum tops occurred during the first harvest suggesting that most decomposition and N mineralization of the legume biomass occurred soon after application. Recovery of ^{15}N in the second sorghum harvest suggested that N mineralized by the legumes was still available during sorghum re-growth. In the same study, hairy vetch decomposed faster than alfalfa, which they attributed to the lower C:N ratio of hairy vetch. A field incubation study by Odhiambo and Bomke (2000) on a silty clay loam in British Columbia showed that a combination of winter wheat and hairy vetch application resulted in net N mineralization through the 16 weeks of study. The most rapid N release occurred within the first two weeks of application. In the same study, application of mixtures of clover (*Trifolium incarnatum* L.), winter wheat and fall rye resulted in net N immobilized between 2 and 4 weeks of incubation. They established that the critical N concentration above which net mineralization occurred was 14 g kg^{-1} .

Ma et al. (1999) used models to predict rates of crop residue decomposition at different slope positions over a 13-year period in a no-till system in eastern Colorado (US). The models indicated air temperature and soil moisture were the main factors influencing decomposition. Crop residue mass loss was determined by collecting 1-m^2 grab samples at different landscape positions at the beginning of the experiment, at planting, and before harvest of each crop. The models assumed that all residues had the

same decay rate as newly added crop residues, and that surface residue of different origin decomposed independently. The models did not predict significant differences in decomposition rates among different slope positions.

Objectives

Soil management and landscape positions (described by terrain attributes) influence greenhouse gas emissions and soil C and N dynamics interactively, rather than in isolation. Evaluation of effects of these interactions on greenhouse gas emissions and soil C and N dynamics is necessary for site-specific management and improving our understanding of these processes. This study evaluates effects of agroecosystem management and landscape variability on: 1) CH₄, N₂O and CO₂ emissions, 2) soil C and N dynamics and 3) decomposition and mineralization of cover crops in a corn -cotton (*Gossypium hirsutum* L.) rotation.

II. AGROECOSYSTEM MANAGEMENT EFFECTS ON GREENHOUSE GAS EMISSIONS ACROSS A COASTAL PLAIN CATENA

ABSTRACT

Soil under crop production may emit trace gases that contribute to climate change through heat-absorbing properties. Topographic variation influences soil properties that influence soil respiration and subsequent trace gas emissions. Among these trace gases are methane (CH₄), nitrous oxide (N₂O) and carbon dioxide (CO₂). Scarcity of data on greenhouse gas emissions as influenced by landscape variability and agroecosystem management in southeastern US necessitates study. The objective of the current study was to evaluate effects of landscape position and agroecosystem management on CH₄, N₂O and CO₂ emissions. Soil management strategies include 1) conventional tillage (CT), 2) conservation tillage (CsT), 3) CT with dairy manure (CTM) and 4) conservation tillage with dairy manure (CsTM) on a corn (*Zea mays* L.)-cotton (*Gossypium hirsutum* L.) rotation. Conservation tillage included white lupin (*Lupinus albus* L.), crimson clover (*Trifolium incarnatum* L.), black oat (*Avena strigosa* Schreb.), and rye (*Secale cereale* L.) cover crops. Each soil management treatment was replicated on summit, sideslope and the drainageway landscape position delineated using both an order 1 soil survey and a digital elevation model (DEM). Seasonal gas measurements were conducted using a closed chamber method from spring 2004 through winter 2006. Results showed that the

drainageway was a CH₄ emitter (emitting 46, 251, 59, and 185mg CH₄-C ha⁻¹ h⁻¹ from CT, CTM, CsT and CsTM treatments, respectively). The summit position was a CH₄ consumer, with fluxes of -59 and -90 mg CH₄-C ha⁻¹ h⁻¹ on CT and CsT treatments, respectively. However, dairy manure application converted the summit landscape to a CH₄ emitter, with 8 and 311 mg CH₄-C ha⁻¹h⁻¹ from CT and CsT, respectively. Averaged across seasons, CT and CsT N₂O fluxes were similar (547 and 437 mg N₂O-N ha⁻¹ h⁻¹, respectively) in the drainageway, the only landscape position in which significant soil management treatment differences on N₂O fluxes were observed. In the drainageway, dairy manure drastically decreased N₂O-N emission on CT treatments (emission of 162 mg N₂O-N ha⁻¹ h⁻¹ on CTM treatments compared to 574 mg ha⁻¹ h⁻¹ N₂O-N from CT treatments). Higher CO₂ fluxes were observed on CsT than on CT treatments in winter seasons. Carbon dioxide emission in winter 2005 from CsT treatments (averaged across landscape positions) was 1304 g CO₂-C, compared to 227 g ha⁻¹ h⁻¹ CO₂-C from CT treatments. Due to complex effects of soil management systems on greenhouse gas (GHG) emissions, agroecosystem management choices should be based on site-specific GHG emission analysis. Adoption of soil management options that promote low GHG emissions should be encouraged, while paying attention to relative ability of the gases to trap heat.

INTRODUCTION

Increase in atmospheric concentrations of greenhouse gases is currently a concern due to their role in climate change. Concentration of the gases in the atmosphere has increased since the beginning of large scale industrialization in the 1750s (IPCC, 2001).

Agriculture alone contributes about 20% of the annual increase in radiative forcing (ability of one metric ton of a greenhouse gas to trap heat relative to a ton of CO₂) through emission of CH₄, N₂O and CO₂ (Cole et al., 1997). An additional 13% annual increase from land clearing via burning raises this contribution to about 33%. To a large extent, emission of these gases depends on agroecosystem management and soil properties. Soil properties are a product of soil forming factors including landscape variability, agroecosystem management and climatic factors. Development and promotion of soil management practices that maximize CH₄ and CO₂ sinks while minimizing N₂O and CO₂ emissions and maintaining crop yields is required.

Carbon dioxide is produced from soil through respiration of plant roots, micro- and macro-flora and fauna, and biochemical oxidation of C containing materials. Tillage is known to influence CO₂ emission from soil (Lee et al., 2006). The magnitude of CO₂ emission from soil due to tillage is highly correlated to intensity of soil disturbance (Reicosky, 1997). Mixing soil during plowing buries surface residues and aerates soil, favoring maximum CO₂ emission owing to increased microbial respiration and CO₂ diffusivity. Inversion tillage results in increased CO₂ emission, with emission levels gradually declining with time (Reicosky, 1997). Thus, time of CO₂ measurement in relation to tillage operations is an important factor in CO₂ measurements.

Methane is second only to CO₂ in its role of producing and enhancing the greenhouse effect (Lowe, 2006). Predominant CH₄ sources include wetlands and digestive activities of ruminant animals (Lowe, 2006). Methane is lost through tropospheric oxidation, stratospheric loss and oxidation in aerobic soils. The oxidation process requires oxygen and is carried out by a diverse group of aerobic bacteria found in

most soils (Meixner and Eugster, 1999). The rate of CH₄ oxidation in soil is influenced by diffusion of the gas to the microorganisms. This diffusion is influenced by water content in the soil and, thus, soil water dynamics are important factors in CH₄ oxidation.

Nitrous oxide results from denitrification, a process that is favored by low oxygen levels in the soil. It also requires readily oxidizable organic C (Meixner and Eugster, 1999). After emission, the gas diffuses to the atmosphere where it may be converted to nitric oxide, a gas known to contribute to depletion of the ozone layer. Oxidation of ammonium through the process of nitrification also produces N₂O as a byproduct of oxidizing bacteria (McSwiney et al. 2001). The process requires oxygen, and therefore soil conditions that favor CH₄ oxidation may favor N₂O emission (through nitrification). According to the Intergovernmental Panel on Climate Change (IPCC, 2001), N₂O is a much more potent greenhouse gas than CH₄. Also, N₂O contributes about 5% of the total greenhouse effect (Pathak, 1999). Soil is known to act mainly as a source of N₂O (as opposed to N₂O sink), although Freney et al. (1978) found some transitory absorption of N₂O under low oxygen concentrations in the atmosphere above the soil under laboratory conditions. However, they found no evidence of N₂O absorption under field conditions. Due to this, emission reduction targets N₂O sources.

Soils vary across landscapes, and the interaction of soil properties with agroecosystem management influences GHG emissions. In regions that receive ample rainfall and have high temperatures, such as the southeastern US, soil C and N dynamics are robust and contribute to greenhouse gases in the atmosphere. However, data on emission of these gases, particularly in relation to landscape variability and

agroecosystem management, are lacking. The objectives of this study were: 1) to compare effects of tillage and dairy manure application and landscape variability on soil CH₄, N₂O and CO₂, and 2) to assess the effect of interactions of landscape variability and soil management on CH₄, N₂O and CO₂ fluxes.

MATERIALS AND METHODS

Study Site

The study site is at the E.V. Smith Research Center near Shorter, AL, US, and lies at 85°53'50''W and 32°25'22''N. The site has a gentle slope ranging from 0-5%, and the soils are classified as Typic, Oxyaquic, and Aquic Paleudults. Details of the surface soil chemical characteristics prior to experiment establishment (2000) at the site have been described by Terra et al. (2006).

Soil Management and Experimental Design

The study site is a 9-ha field containing a corn (*Zea mays* L.)-cotton (*Gossypium hirsutum* L.) rotation. Soil management treatments were established in 6.1 m wide by ~240 m long strips across the landscape (Fig. 1) in a randomized complete block design with six replications. Plots measuring 6.1 m x 18.3 m were delineated in each strip, resulting in a total of 496 plots. Soil management treatments implemented in fall 2000 include: 1) conventional tillage (CT) involving disking, chisel plowing, field cultivation (to level seedbed), 2) conventional tillage + dairy manure (CTM) applied each fall at a rate of ~ 10 Mg ha⁻¹ (fresh weight basis), 3) conservation tillage (CsT) consisting of non-inversion in-row subsoiling and winter cover crops of white lupin (*Lupinus albus* L.) and

crimson clover (*Trifolium incarnatum* L.) prior to corn and rye (*Secale cereale* L.)/black oat (*Avena strigosa* Schreb.) mixture prior to cotton and 4) conservation tillage + dairy manure (CsTM) applied in the fall at a rate of $\sim 10 \text{ Mg ha}^{-1}$. Further details on experiment treatments can be found in Terra et al. (2006).

The field was divided into three soil landscape positions (Fig. 1) using an order 1 soil survey (1:5000) and a high resolution digital elevation model (DEM). Digital elevation data were obtained using a Real Time Kinematic (RTK)-GPS. Elevation data were interpolated to provide a DEM in Arc Info (ESRI, Redlands, CA), and used to develop slope and the compound topographic index (CTI) (Moore et al. 1993). The compound topographic index relates specific catchment area to slope. Soil survey data were rasterized to indicate seasonal high water table (SHWT) and overlaid with DEM, slope and CTI layers. Fuzzy k-means unsupervised clustering of these multivariate data was used to delineate three landscape positions (summit, sideslope and drainageway) (Fridgen et al., 2004).

In spring 2004, 36 GPS referenced plots were identified for trace gas measurements. Plots were distributed across the three landscape positions and four management systems cropped to cotton during 2004. These plots were under corn rotation in 2005 and under cotton in 2006. Each management treatment was replicated three times ($3 \times 4 \times 3 = 36$ plots).

Dairy manure was applied on October 22, 2004 and November 19, 2005. On CT plots, disking and plowing were performed on April 29, 2004 and April 5, 2005. The dairy manure applied in fall 2004 had total P, K, Ca, and Mg of 3.4, 1.3, 29 and 8.9 g kg^{-1} manure as determined through nitric/perchloric acid wet ashing (Hue and Evans, 1986)

while total N (Kjedahl digestion) was 8.2 g kg⁻¹ manure. In fall 2005, the dairy manure had total P, K, Ca, and Mg of 0.9, 0.9, 7.8 and 2.1 g kg⁻¹ manure respectively, while N content was 6.2 g kg⁻¹ manure. Dairy manure moisture content was 44% and 70% in 2004 and 2005, respectively.

Gas Measurement

Gas sample measurements were taken once every season for a period of two years using the static closed chamber method described by Mosier and Schimel (1991). Gas samples were obtained on May 12, 2004, August 5, 2004, October 27, 2004, January 20, 2005, April 29, 2005, July 22, 2005, November 7, 2005 and January 26, 2006. Chambers were constructed from 20 cm diameter PVC pipes and were 16 cm in height. They comprised a lower base and an upper detachable cap with top surface lined with reflective foil to maintain ambient air temperature in the chamber headspace. The bottom edge was sharpened to facilitate chamber installation and prevent soil compaction. The cap was fitted with a 5 mm diameter vent and a removable gray butyl rubber septum sampling port. A day prior to gas sample collection, the chamber base was pushed into the soil to a depth of 3 cm, leaving the rest of the chamber above the soil surface and open to the atmosphere. Chambers were placed on the middle non-trafficked parts of the plot. At start of gas sample collection, chamber caps were placed on each base and held in place with a latex elastic band. Gas was sampled at 30 minute intervals for a period of one hour.

In order to represent daily average temperatures at the site, gas samples were taken during the mid-morning. Three mL of gas were collected from the chamber headspace using a 3 mL disposable syringe equipped with a needle. In order to ensure a

representative sample from the chamber, the syringe was pumped three times to mix the gas in the chamber headspace before taking out a sample. Samples were transferred to 3 mL glass storage vials, stored at 4°C, and transported to the laboratory where they were stored at the same temperature until analysis. Prior to gas sampling, storage vials were capped with gray butyl rubber septa at the gas sampling site to ensure similar background conditions in the vials and the sampling site. At each sampling time, two samples were obtained. One sample was used for CH₄ determination while the other was for N₂O and CO₂ analyses. Gas samples were analyzed using a Varian Star *cx* gas chromatograph (Varian, Walnut Creek, CA). Nitrous oxide and CO₂ were determined (from one vial) using a 4 m Haysep R column and a ⁶³Ni electron capture detector (ECD). The detector temperature was 350°C, and the carrier gas was N₂ (17 mL min⁻¹ flow rate). Methane concentrations were determined using a 3 m Porapak N column and a flame ionizing detector (FID). The detector temperature was 350°C and the carrier gas was N₂ at a flow rate of 30 mL min⁻¹. Calibration curves were generated using respective gas standard samples and CH₄, N₂O and CO₂ fractions (by volume) were calculated from the peak area in the chromatograms.

At each gas sampling time, soil temperature was determined on one plot per replication using HOBO[®] Temperature Probes (Forestry Supplies Inc. Jackson, MS).

Gas Flux Calculations

Gas flux calculations were based on chamber volume and soil surface area covered by the chamber. Gas volume at standard temperature and pressure was assumed in the calculations (22.4 L mole⁻¹). Chamber head space internal volume above the soil

surface was 4.08 L calculated from a chamber diameter of 0.2m and a height of 0.13 m above the soil surface. Chamber volume occupied by each gas was calculated from the gas concentration obtained from the gas chromatography analysis, and subsequently used to determine the number of moles of each gas in the chamber at time of sampling using the ideal gas law. This was further converted to mass of C in the case of CH₄ and CO₂, and N for N₂O, and expressed on soil area basis. Gas flux was determined by linearly regressing time of gas accumulation against respective mass per unit area.

Soil Sampling and Analysis

During each gas sampling time, soil samples were obtained from 0-5 cm depth using a 2.0 cm diameter hand probe. On each plot, 20 samples were obtained in a random manner and combined to form one composite sample per plot. Samples were stored at 4°C until analysis for mineral N (NH₄ and NO₃-N), organic C and total N. Gravimetric soil moisture content was determined by drying 1 g soil at 105° to constant weight. Mineral N was determined by extraction with 2M KCl at a ratio of 1:5 (soil:KCl) and concentrations of NH₄ and NO₃ were determined colorimetrically using a μQuant™ micro-plate spectrophotometer (BioTek instruments, Inc. Winooski, VT). Organic soil C and total N were determined using LECO TruSpec CN analyzer (Leco Corp., St. Joseph, MI).

Data Analysis

The generalized linear model (GLM) in SAS (SAS Inst., Cary, NC) was used to compare terrain attributes across landscape positions. Treatment means were compared using Fisher's protected least significant difference (LSD) at $P \leq 0.05$.

For CH₄, N₂O and CO₂ fluxes, the mixed generalized linear model using PROC MIXED in SAS was used to account for repeated measures across seasons, and to test for main effects and interactions. Treatment means were compared using least significant difference (LSD) calculated from standard errors obtained from the PROC MIXED procedure. Additionally, paired t-tests were used to compare the effect of tillage and dairy manure on CH₄, N₂O and CO₂ fluxes.

Stepwise regression was used to relate terrain attributes to gas emissions in seasons when significant soil management treatment (tillage and dairy manure) effects were observed. Terrain attributes used in the regression analysis include CTI, digital elevation, slope, planimetric curvature, profile curvature, flow accumulation, SHWT and surface horizon sand, silt and clay content as determined by Terra et al. (2006).

Methane, N₂O and CO₂ flux data were normalized (0-100) followed by principal component analysis in SAS. Methane data was for spring 2004, while N₂O and CO₂ was for eight seasons starting in spring 2004 through winter 2006. A plot of scores of the first two principal components was utilized to determine if there were distinct groupings relating to landscape and soil management treatments.

RESULTS

Landscape Variability

There were significant differences in landscape variability factors between landscape positions, except planimetric curvature and surface horizon silt content (Table 1). Sideslope landscape position had higher slope, surface horizon clay content and profile curvature compared to the drainageway position. Higher slope on the sideslope result in runoff that would accumulate in the drainageway. This is depicted by higher flow accumulation, surface horizon sand content and compound topographic index (CTI) within the drainageway landscape. Positive profile curvature values found on the summit landscape position indicate a convex profile, while negative profile curvatures on drainageway indicate concave profiles (Li et al., 2005). Highest elevations and depth to seasonal high water table (SHWT) were found on the summit.

Methane Fluxes

Samples for CH₄ flux determination were collected seasonally between spring 2004 and winter 2006, but only spring 2004 fluxes are reported due to gas chromatography CH₄ channel failure in subsequent seasons. Some trends in soil management impacts on CH₄ flux ($P = 0.312$) were discernable (Fig. 2). The summit landscape position under CT and CsT treatment was a CH₄ sink, while the drainageway emitted CH₄ (as shown by negative and positive fluxes respectively) (Fig. 2). Fall dairy manure application converted the summit landscape position from a CH₄ sink to a CH₄ producer the subsequent spring (Fig. 2). Generally, dairy manure increased CH₄-C production except on the sideslope landscape position, where minimal negative fluxes

were observed on CsTM treatments. Comparison (t-test) of soil management treatment means at the landscape level showed higher CH₄ fluxes on CsTM treatments compared to CT treatments on the summit and in the drainageway (Table 2). Although not statistically different, CH₄-C consumption rate on CsT treatments was about twice that on CT treatments on the summit landscape position. On the same landscape position, CTM had an average of 8 mg CH₄-C ha⁻¹ h⁻¹ while CsTM had an average flux rate of 310 mg CH₄-C ha⁻¹ h⁻¹. The positive CH₄ fluxes in the drainageway were in the order CTM > CsTM > CsT > CT.

Nitrous Oxide Fluxes

A soil management by season interaction (P = 0.031) resulted in N₂O flux differences in spring 2004 and fall 2005 (Fig. 3 a and b). In spring, CT and CsT had similar fluxes, but CTM had higher fluxes than CsT and CsTM treatments. In the fall, CsT had greater N₂O fluxes than CT treatment. Dairy manure decreased N₂O flux on CsT treatments (CsT and CsTM). In both seasons, terrain attribute effects on N₂O fluxes varied with soil management (Table 3). In spring 2004, slope had a negative effect on N₂O flux on CTM treatments. In fall 2005, landscape variability had no effect on N₂O fluxes on the CsT treatment. However, surface horizon clay content explained 71% of N₂O flux variability on CTM treatments.

Soil management interacted with landscape position (P = 0.037) to affect N₂O-N fluxes. Significant soil management treatment differences in N₂O-N flux were observed only in the drainageway (Fig. 3 c). Averaged across soil management treatments and seasons, average N₂O-N flux in the drainageway was 346 mg ha⁻¹ h⁻¹ N₂O-N relative

to 158 and 220 mg ha⁻¹ h⁻¹ N₂O-N on the summit and sideslope, respectively. Within the drainageway, no N₂O-N flux differences were observed between the two tillage systems but, fluxes were higher on CT than on CTM and CsTM treatments (Fig. 3 c). Thus, within the CT system, dairy manure application (CTM) decreased N₂O flux, while it had no significant effect on CsT system fluxes.

A significant season by landscape position interaction (P = 0.002) indicate that N₂O flux differences occurred in spring and fall 2004 (Fig. 4). In spring, highest fluxes were observed on the summit landscape position, while in fall, higher fluxes were in the drainageway.

Carbon Dioxide Emission

Season and soil management interacted to alter CO₂ emission (P = 0.001). Significantly different CO₂ emissions were observed in winter 2005 (Jan-05), during which CsT treatments had higher emission (1304 g ha⁻¹ h⁻¹ CO₂-C) than CT treatments (227 g ha⁻¹ h⁻¹ CO₂-C) (Fig. 5). A similar trend was observed in winter 2006 (Jan-06) when CsT emitted 1151 g ha⁻¹ h⁻¹ CO₂-C, compared to 390 g ha⁻¹ h⁻¹ CO₂-C on CT treatments. Although not significantly different, higher CO₂ emissions were observed on CTM treatments compared to CT treatments during both winter seasons. Conservation tillage + dairy manure treatments had lower CO₂ emission than CsT treatments, but the difference was not significant in winter 2005.

Effect of terrain attributes on winter CO₂ emission depended on soil management (Table 4). In winter 2005, surface horizon sand content had a positive influence on CO₂ emission on CTM treatments, and 83% of the emission variation could be attributed to

this factor. Flow accumulation explained 71% of CO₂ emission on CT treatments, while no factor contributed significantly to CO₂ emission on CsTM treatments. In winter 2006, the main terrain attribute influencing CO₂ emission was slope. Slope explained 29% and 28% of CO₂ emission variability on CT and CsT treatments, respectively.

Soil Variables

There were significant season by landscape position by soil management interactions for soil NH₄-N ($P = 0.001$). In spring 2004, higher soil NH₄-N was observed in CT treatments on the summit and sideslope landscape position (Fig. 6). In the same season, addition of dairy manure (CsTM) on conservation tillage within the drainageway reduced soil NH₄-N from 4.9 to 0.7 mg kg⁻¹ soil (Fig. 6). In summer 2004, significant soil management treatment differences were observed only on the sideslope, where higher soil NH₄-N was observed on CsT treatments compared to CTM treatments (Fig. 6). No soil management treatment differences were observed in fall 2004 in all landscape positions. Higher NH₄-N was observed on CT treatments on summit and drainageway landscape positions in winter 2005, while higher NH₄-N was observed on CsT in sideslope position in the same season. In spring 2005, significant differences in soil NH₄-N occurred only on the sideslope position and were in the order CsTM = CsT > CTM = CT. Similar to summer 2004, no soil management NH₄-N differences were observed in summer 2005 on summit and drainageway landscape positions, but CsT had higher NH₄-N on sideslope landscape position. Relatively higher amounts of soil NH₄-N were observed in fall (Oct.) 2005 (Fig. 6), with higher NH₄-N observed on CsT treatments on summit and sideslope landscape positions. In winter 2006, CsTM treatments on the

sideslope landscape position had higher $\text{NH}_4\text{-N}$ than any other soil management landscape combination. On the same landscape position, CTM treatment had twice as much $\text{NH}_4\text{-N}$ as CT treatment.

Significant season by soil management ($P = 0.001$) interaction effects on soil $\text{NO}_3\text{-N}$ were observed in all seasons except spring 2004, summer 2004 and winter 2005 (Fig.7). In fall 2004, CsT treatments yielded higher $\text{NO}_3\text{-N}$ than CT treatments, while dairy manure increased $\text{NO}_3\text{-N}$ levels in both tillage systems (Fig. 7). Highest seasonal $\text{NO}_3\text{-N}$ levels were recorded in spring of 2005 with CsT treatments showing higher $\text{NO}_3\text{-N}$ compared to CT treatment. In spring 2005, dairy manure did not significantly affect $\text{NO}_3\text{-N}$ levels in either tillage system. Similar trends were observed in winter 2006.

There were significant season by soil management interaction effects ($P = 0.001$) on total soil C. Conservation tillage showed higher soil organic C (averaged across landscape positions) compared to CT in each season except in summer 2004. Dairy manure increased soil organic C in both tillage systems in all seasons except winter 2005 when dairy manure had no significant effect on soil organic C on CT (Fig. 8). Within the CT treatments, total organic C was more or less constant throughout the two years. Averaged across all seasons, CsT treatments had 13.1 g C kg^{-1} soil compared to 7.6 g C kg^{-1} soil on CT treatments.

There were significant seasonal differences ($P = 0.001$) and landscape position by soil management treatment interactions ($P = 0.049$) for total soil N. In summer 2004 and winter 2005, higher total N was found on summit landscape relative to the drainageway and sideslope position (Fig. 9). In all seasons, CsTM treatment had the highest total soil N, while CT had the lowest levels (Fig. 10).

There were significant season by landscape position ($P = 0.003$) and season by soil management ($P = 0.001$) interactions on gravimetric soil moisture (0-5 cm). In all seasons, consistently higher soil moisture was found on CsTM treatments, while the lowest moisture levels were found on CT treatments (Fig. 11 a). Additionally, dairy manure increased soil moisture in both tillage systems. Higher water content was found in the drainageway, while similar water contents were observed on the summit and the sideslope landscape positions (Fig. 11 b). These differences were observed in fall 2004, winter 2005 and winter 2006.

Gas Flux Multivariate Analysis

A plot of the scores of the first two principal components is shown on Fig. 12. These components described 49% of the normalized data variability. In general, CsT treatments fall below the zero line, while CT treatments fall above the zero line. Thus, principal components separated the gas flux data based on tillage system regardless of landscape position and dairy manure treatments.

DISCUSSION

Methane Fluxes

Methane exhibited negative as well as positive fluxes, with no differences due to tillage. Although reduced tillage has been observed to increase CH_4 consumption by minimizing soil disturbance favorable to CH_4 oxidizing bacteria (Hütsch, 1997), CT did not reduce CH_4 consumption significantly compared to CsT treatments (Appendix 2). Hütsch (1997) found that sieving intact soil cores (5-mm) reduced CH_4 oxidation by 57%

and 15% on sandy and loamy soils, respectively. Lower CH₄ oxidation on sandy soil was attributed to greater destruction of soil aggregates that reduce methane oxidizing bacteria. Similar to our findings, Suwanwaree and Robertson (2005) found no effect of plowing on CH₄ oxidation along a management intensity gradient ranging from virgin forest to a no-till corn-soybean-wheat rotation in Michigan.

Other factors such as soil moisture and temperature influence CH₄ fluxes. Laboratory studies have shown an optimum methane oxidation temperature of 20-30°C (Boeckx et al., 1996). This optimum temperature decreased with increasing soil moisture. In our study, mean soil temperatures on both tillage systems were similar (27-31°C and 29-32°C on CT and CsT treatments, respectively) and may, in part, explain lack of CH₄ flux differences in the two systems. Soil moisture was also similar in both systems. Similarly, Chan and Parkin (2001) found no difference in fluxes between no-till and plowed sites. They attributed this to field spatial variation, but in our study spatial variation was largely accounted for by stratifying the plots by landscape.

According to Venterea et al. (2005), the effect of tillage on CH₄ emissions depends on the type of N fertilizer used. In their study in Minnesota, urea ammonium-nitrate resulted in no differences in CH₄ emissions between tillage systems, whereas urea increased CH₄ emission on reduced tillage systems. In our study, no N fertilizer had been applied prior to gas measurements other than that applied to corn in the previous cropping season, and 34 kg N ha⁻¹ ammonium-nitrate applied to CsTM treatments three months earlier. It is important to note that tillage operations were done in early spring, while gas measurements were 13 days later. Effect of tillage on factors that control CH₄ fluxes may have diminished with time following cultivation.

Landscape variability influences CH₄ fluxes due to accompanying differences in soil properties that interact with soil management. This may result in different CH₄ flux responses at short distance intervals (local-scale variability) within a landscape. In the current study, higher soil moisture in the drainageway appears to have favored CH₄ production, resulting in net CH₄ emission in this landscape position. Soil temperature in the drainageway ranged between 27-30°C. According to Meixner and Eugster (1999), most CH₄ producing bacteria operate within a temperature range of 20-40°C. Similar results were found by Chan and Parkin (2001) when they measured CH₄ fluxes along a transect traversing a field in Iowa, (US). Low lying areas gave positive CH₄ fluxes, while higher areas had negative CH₄ fluxes.

The main substrate in CH₄ production in soil is acetate and results from fermentation of several substances including organic matter (Meixner and Eugster, 1999). Decomposition of dairy manure may provide this raw material for CH₄ production, and may explain greater CH₄ fluxes on dairy manure treatments under CT and CsT.

Nitrous Oxide Fluxes

Nitrous oxide flux varied with seasons and landscape position (Appendix 2). Summit and sideslope landscape positions did not show soil management differences, perhaps due to similar soil moisture between soil management. Soil management differences were observed in the drainageway, with CT treatments showing higher fluxes than CTM and CsTM treatments. The drainageway tended to have higher soil moisture in all seasons (Fig. 11 b). Conversely, seasonal soil management treatment differences in N₂O-N flux observed in spring 2004 and fall 2005 correspond with seasons that had

lower soil moisture (Fig. 11 a) and higher soil $\text{NH}_4\text{-N}$ (Fig. 6). Other than summer 2004 when soil moisture was extremely low, spring 2004 and fall 2005 had the lowest gravimetric moisture contents. Highest soil $\text{NH}_4\text{-N}$ levels over the entire study period were measured during these two seasons (Fig. 6), suggesting the N_2O measured was mainly a result of NH_4 nitrification. During the two seasons, N_2O fluxes were influenced by soil NH_4 as indicated by the similarity between N_2O flux (Fig. 3 c) and soil NH_4 trends (Fig. 6). Similarly, Breuer et al. (2002) found positive correlation between nitrification and N_2O emission, and negative correlation between nitrification and increasing rates of water-filled porosity.

Lack of a significant tillage effect on N_2O flux within landscape positions may be due to similar soil moisture and temperature between the tillage systems. Mean soil temperature (averaged across seasons) on each individual landscape position was between 20-23°C. Higher mean N_2O flux on CsT compared to CT in fall 2005 (Fig. 3) may be related to the relatively higher NH_4 (Fig. 6) and NO_3 (Fig. 7) on these treatments compared to CT treatments in this season. Nitrous oxide is a product of NH_4 nitrification and denitrification of NO_3 (Meixner and Eugster, 1999). Both processes are controlled by oxygen concentration, but McSwiney et al. (2001) pointed out that high N_2O concentration in a location could be a result of gas production or gas accumulation.

Notably, CT treatments consistently had the lowest gravimetric soil moisture (0-5 cm) each season, while CsTM treatments had the highest soil moisture levels (Fig. 11 a). Though not significantly different, the opposite trend was observed on N_2O flux (Fig. 3 c). Thus, N_2O fluxes appear to negatively correlate with soil moisture, although the differences in these levels may not have been sufficient to result in significant soil

management treatment differences. High soil moisture conditions can create reducing conditions where N_2O is reduced to N_2 . Low N_2O flux in winter 2005 and 2006 may be a result of lower soil temperatures (a seasonal average of $9.8^\circ C$ and $9.0^\circ C$ respectively) and relatively high soil moisture. Dairy manure tended to increase soil moisture levels and resulted in a decrease in N_2O flux. Dairy manure significantly decreased N_2O flux within the drainageway in both tillage systems. This too may be attributed to accompanying increase in soil moisture.

In spring 2004 and fall 2005 when significant soil management treatment (tillage and dairy manure) effects on N_2O fluxes were observed within the drainageway, landscape variability effects on N_2O flux were not consistent in the two seasons. Positive relationship between surface horizon sand content (Table 3) and N_2O fluxes on CT treatments in spring 2004 is consistent with negative correlation between N_2O fluxes and soil moisture content during this season. Soils with high sand content generally have low amounts of available moisture. However, in fall 2005, no single terrain attribute could reasonably explain N_2O flux variance on CsT treatments. High surface horizon clay content resulted in decreased N_2O fluxes on CTM treatments, perhaps due to its positive influence on soil moisture. Variation in effect of terrain attributes on N_2O fluxes across seasons may not be surprising given that terrain attributes act interactively with environmental factors in their influence on microbial activities. Whereas terrain attributes may not change much over short time periods, environmental factors are dynamic and a change in these factors is reflected accordingly in soil microbial activities.

Carbon Dioxide Emission

We observed higher CO₂ emission under CsT than on CT treatments in winter, with no soil management treatment differences in other seasons. The reason for this observation is not clear, as soil temperature and moisture levels were comparable in both systems. It may be due to differences in gas diffusivity in the two systems as a result of differences in soil porosity. According to Hashimoto and Komatsu (2005), CO₂ flux is a function of CO₂ respiration and diffusivity. Soils managed under conservation tillage may be more porous due to annual addition of winter cover crop residues. Total soil C and N were similar in both systems, and the resulting soil C:N ratio ranged from 9 -15, levels at which net mineralization (with subsequent CO₂ release) would be expected. As expected, CO₂ fluxes were lowest in winter (on conventional tillage systems) and may be associated with low winter soil temperatures. Low fluxes observed in summer 2004 (August 2004) may be related to noticeably low soil moisture (Fig. 11 a) and high soil and air temperature (data not shown).

On CT treatments, CO₂ emission was positively influenced by factors that favor increased soil moisture. Flow accumulation increased CO₂ emissions, while slope had a negative effect on CO₂ emissions. Lowest slopes were found in the drainageway landscape position that also had higher soil moisture. The effect of slope on CO₂ emission depended on soil management. Whereas higher slope favored CO₂ emissions on CsTM treatments in winter 2006, it also negatively influenced emissions on CsT treatments in the same season. This suggests a delicate balance exists between soil management and terrain attribute effects on CO₂ emissions.

Gas Flux Principal Component Analysis

Most gas emission variability (49%) was explained by the first two principal components. Using the first two principal components, gas emission may be categorized into two groups (Fig. 12). The two groups are based on tillage system irrespective of landscape position or dairy manure application. This suggests that at our site, tillage had greater impact on CH₄, N₂O and CO₂ fluxes than terrain attributes or dairy manure application.

CONCLUSIONS

Spatial variability is an important factor in site-specific management for environmental protection. Climate change may be somewhat mitigated by soil management strategies that minimize GHG emissions. Knowledge of landscape variability effects on these emissions is important as the magnitude of emissions of different trace gases is influenced by soil and environmental factors.

In this study, no tillage differences were observed for CH₄ and N₂O fluxes. However, CsT treatments emitted higher CO₂ than CT treatments during winter. Low lying areas (drainageway) were CH₄ emitters, and the addition of dairy manure magnified CH₄ emissions in the drainageway. The summit landscape was a CH₄ consumer, but dairy manure application converted it into a CH₄ emitter. Similarly, higher N₂O flux in CT than in CTM and CsTM were observed only in the drainageway, while no landscape effect was observable for CO₂ fluxes. However, seasonal variations were eminent on both N₂O and CO₂ fluxes. Nitrous oxide fluxes followed soil NH₄ trends, and varied

with season. Generally, CH₄, N₂O and CO₂ fluxes were more influenced by tillage than terrain attributes as shown by principal component analysis.

The results of this study could be used in modeling efforts with the goal of predicting GHG emissions as influenced by landscape and management factors.

Table 1. Analysis of variance of terrain attributes among summit, drainageway and sideslope landscape positions at E.V. Smith Research Center near Shorter, AL.

Terrain attribute	Soil landscape position†			P-value
	Summit	Sideslope	Drainageway	
CTI‡	3.98b	4.15b	6.16a	0.0001
Elevation (m)	71.33a	70.53b	69.49c	0.0001
Planimetric curvature	0.01a	-0.01ab	-0.08b	0.0620
Profile curvature	0.02a	0.02a	-0.09b	0.0030
Slope (%)	0.60c	3.33a	1.33b	0.0001
Flow accumulation	05.01b	7.13b	30.35a	0.0040
SHWT (cm)§	145.83a	108.33b	75.00c	0.0010
Sand (%)¶	56.78b	54.32b	63.75a	0.0001
Silt (%)	24.44a	25.50a	25.20a	0.7360
Clay (%)	18.79a	21.12a	11.06b	0.0001

† Values in rows followed by the same letter are not significantly different at $P \leq 0.05$.

‡ Compound topographic index

§ Seasonal high water table

¶ Surface horizon sand, silt and clay content

Table 2. Effect of soil management on CH₄ fluxes on three landscape positions at E.V. Smith Research Center near Shorter, AL.

Soil management treatments †	n‡	Mean CH ₄ flux difference		
		Summit	Sideslope	Drainageway
—————mg CH ₄ -C ha ⁻¹ h ⁻¹ —————				
CT-CTM	3	-67	-126	-205
CT-CsT	3	31	-231	-13
CT - CsTM	3	-370	-39	-139
CTM - CsT	3	98	-106	192
CTM- CsTM	3	-303	87	66
CsT - CsTM	3	-400	192	-126
<u>T-test</u>				
—————P-value—————				
CT-CTM		0.543	0.202	0.205
CT-CsT		0.806	0.342	0.909
CT - CsTM		0.001	0.273	0.013
CTM - CsT		0.065	0.487	0.031
CTM- CsTM		0.074	0.411	0.577
CsT - CsTM		0.064	0.453	0.312

† CT, conventional tillage; CTM, conventional tillage with dairy manure; CsT, conservation tillage; CsTM, conservation tillage with dairy manure.

‡ Sample size

Table 3. Stepwise regression relating terrain attributes to N₂O flux. Only variables with the highest significant contribution to flux variability in each soil management treatment are shown. A positive sign (+) indicates that an increase in the given variable causes an increase in N₂O flux, while a negative (-) sign indicates the opposite.

Season	Soil management treatment†	Independent variable‡	Partial R ²	P-value
Spring 2004	CT	Sand (+)	0.393	0.071
	CTM	Slope (-)	0.459	0.045
	CsT	SHWT (+)	0.478	0.039
	CsTM	SHWT (+)	0.559	0.021
Fall 2005	CT	CTI (+)	0.485	0.037
	CTM	Clay (-)	0.709	0.004
	CsT	None	NS#	NS
	CsTM	Profile curvature (+)	0.295	0.131

† CT, conventional tillage; CTM, conventional tillage with dairy manure; CsT, conservation tillage; CsTM, conservation tillage with dairy manure.

‡ SHWT, seasonal high water table; CTI, compound topographic index; sand, surface horizon sand content; clay, surface horizon clay content.

Not significant at ≤ 0.15

Table 4. Stepwise regression relating terrain attributes to CO₂ flux. Only variables with the highest significant contribution to flux variability in each soil management treatment are shown. A positive sign (+) indicates that an increase in the given variable causes an increase in CO₂ flux, while a negative (-) sign indicates the opposite.

Season	Soil management treatment [†]	Independent variable [‡]	Partial R ²	P-value
Winter 2005	CT	Flow accumulation (+)	0.710	0.004
	CTM	Sand (+)	0.834	0.001
	CsT	Clay (-)	0.400	0.070
	CsTM	None	NS#	NS
Winter 2006	CT	Slope (-)	0.286	0.138
	CTM	Silt (+)	0.370	0.083
	CsT	Slope (-)	0.276	0.147
	CsTM	Slope (+)	0.375	0.080

[†] CT, conventional tillage; CTM, conventional tillage with dairy manure; CsT, conservation tillage; CsTM, conservation tillage with dairy manure.

[‡] Sand, surface horizon sand content; clay, surface horizon clay content; silt, surface horizon silt content.

Not significant at ≤ 0.15

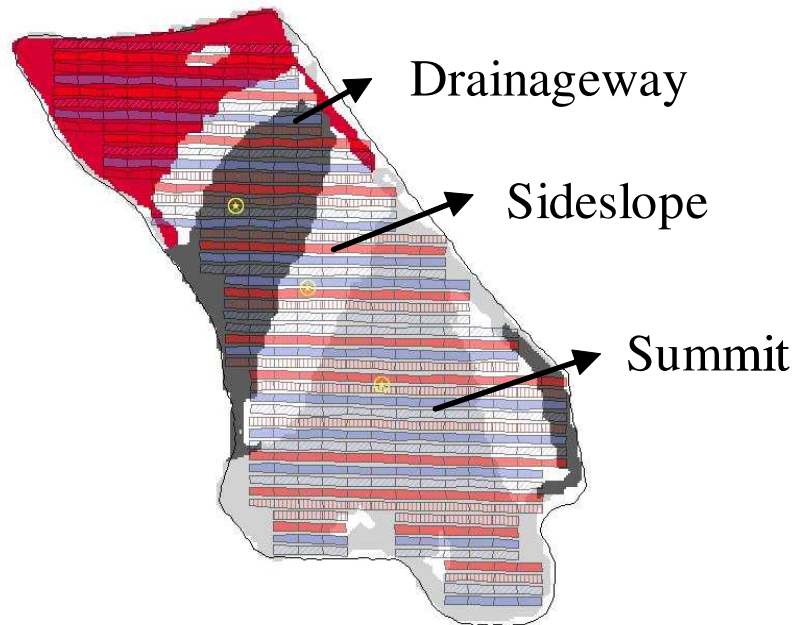


Fig. 1. Study site soil landscape positions created using fuzzy k-means unsupervised clustering based on seasonal high water table, digital elevation, slope and compound topographic index. Summit is the highest position, drainageway the lowest position, while sideslope is an eroded landscape.

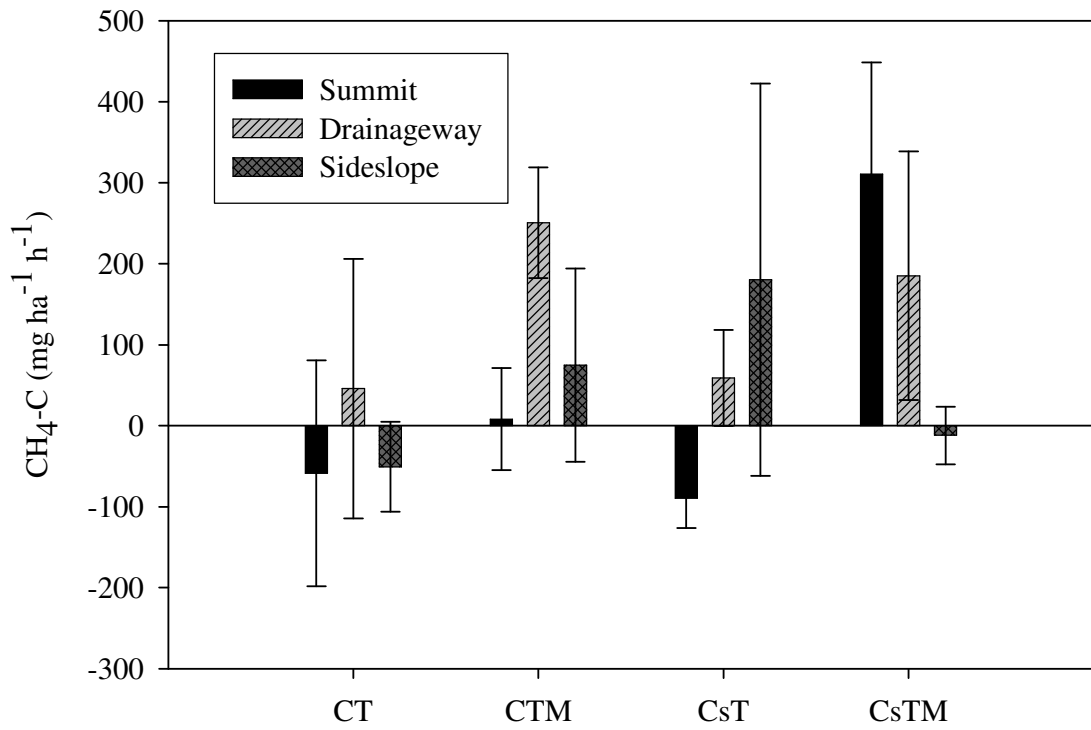


Fig. 2. Methane fluxes on three soil landscape positions at E.V. Smith Research Center near Shorter, AL. CT = Conventional tillage, CsT = Conservation tillage, CTM = Conventional tillage + dairy manure, CsTM = Conservation tillage + dairy manure. Bars are standard errors of the mean.

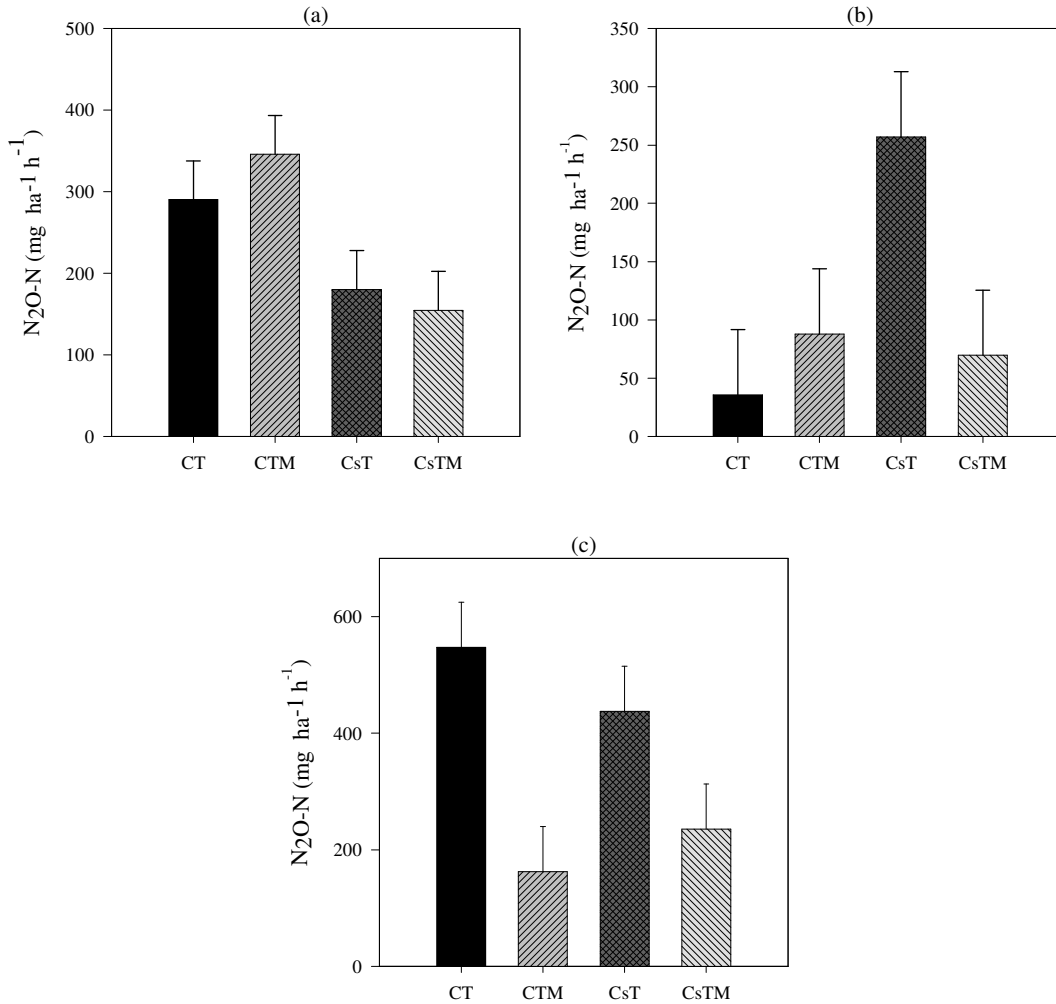


Fig. 3. Soil N₂O fluxes from summit, sideslope and drainage way landscape positions in (a) spring 2004, and (b) fall 2005. (c) represents mean fluxes from drainage way landscape averaged across seasons. CT = Conventional tillage, CsT = Conservation tillage, CTM = Conventional tillage + dairy manure, CsTM = Conservation tillage + dairy manure. Bars represent standard errors of the mean.

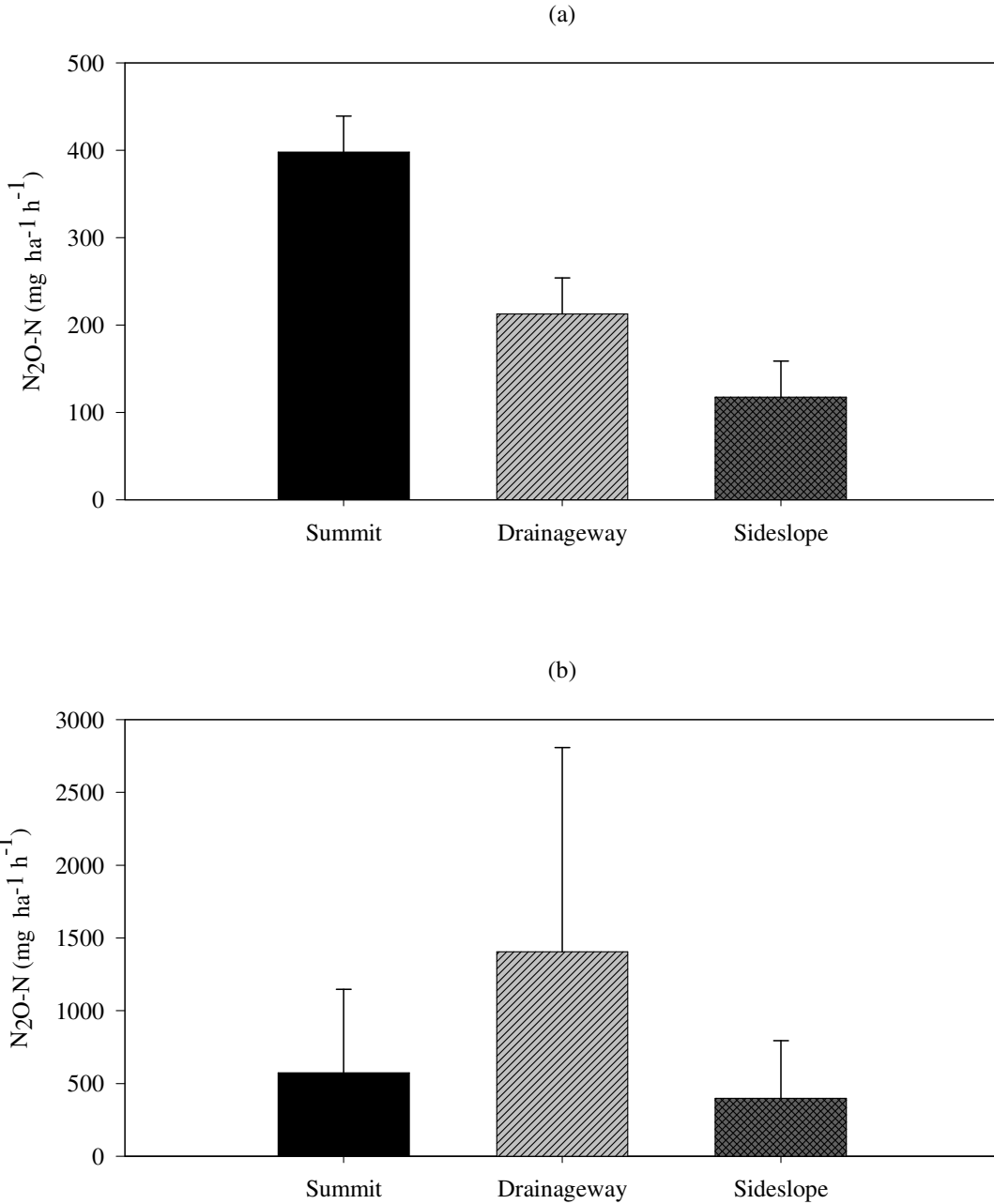


Fig. 4. Soil N₂O flux as influenced by landscape position and season in (a) spring 2004 and (b) fall 2004. Bars are standard errors of the mean.

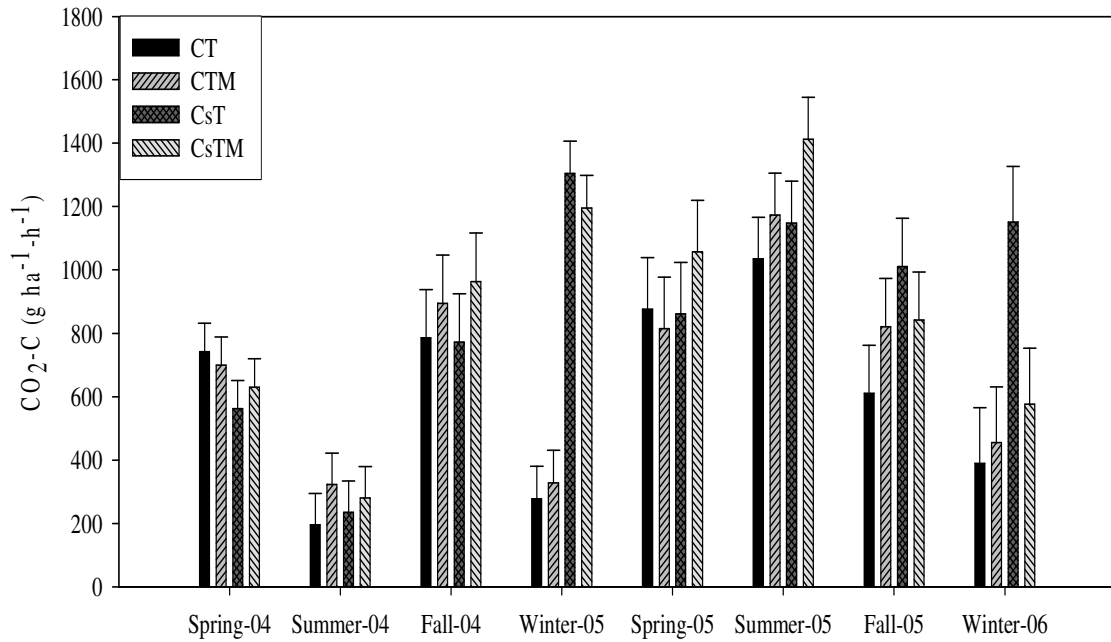


Fig. 5. Seasonal changes in CO₂ fluxes due to tillage and dairy manure application. Data are means from summit, sideslope and drainageway landscapes. CT = Conventional tillage, CsT = Conservation tillage, CTM = Conventional tillage + dairy manure, CsTM = Conservation tillage + dairy manure. Bars are standard errors of the mean.

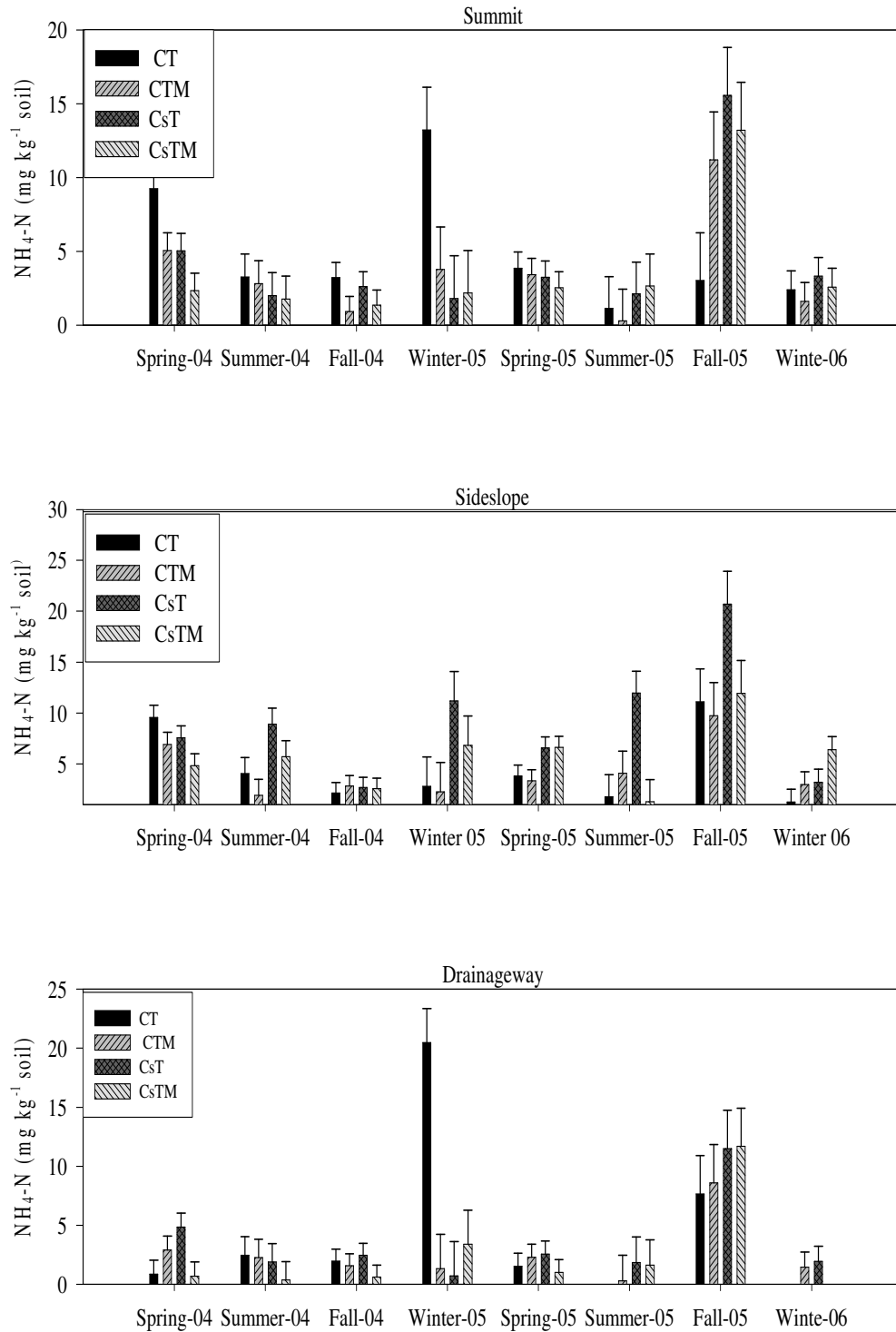


Fig. 6. Soil $\text{NH}_4\text{-N}$ seasonal variation as affected by landscape position and soil management. CT = Conventional tillage, CsT = Conservation tillage, CTM = Conventional tillage + dairy manure, CsTM = Conservation tillage + dairy manure. Bars are standard errors of the mean.

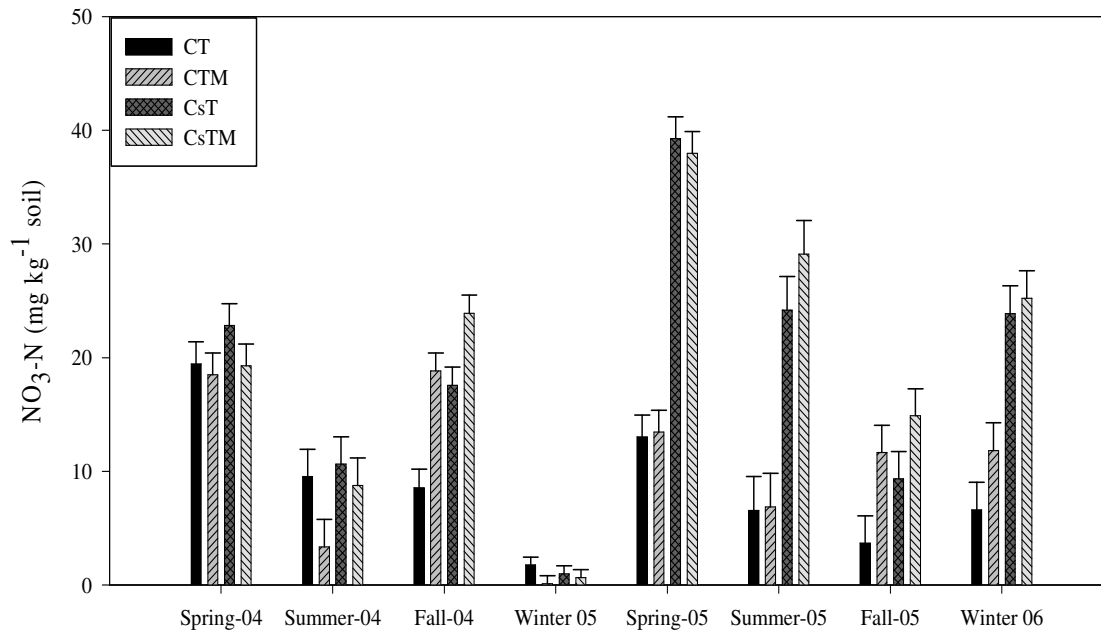


Fig. 7. Seasonal soil NO₃-N changes due to tillage and dairy manure application averaged across three landscape positions. CT = Conventional tillage, CsT = Conservation tillage, CTM = Conventional tillage + dairy manure, CsTM = Conservation tillage + dairy manure. Bars are standard errors of the mean.

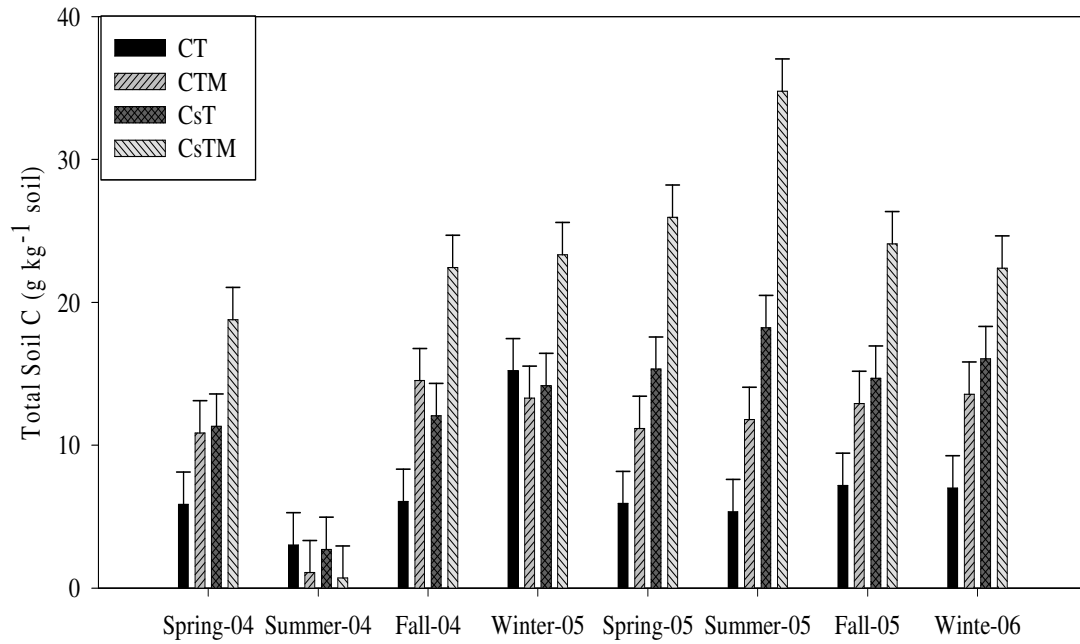


Fig. 8. Organic soil C changes under conventional and conservation tillage systems with and without dairy manure application. CT = Conventional tillage, CsT = Conservation tillage, CTM = Conventional tillage + dairy manure, CsTM = Conservation tillage + dairy manure. Bars are standard errors of the mean.

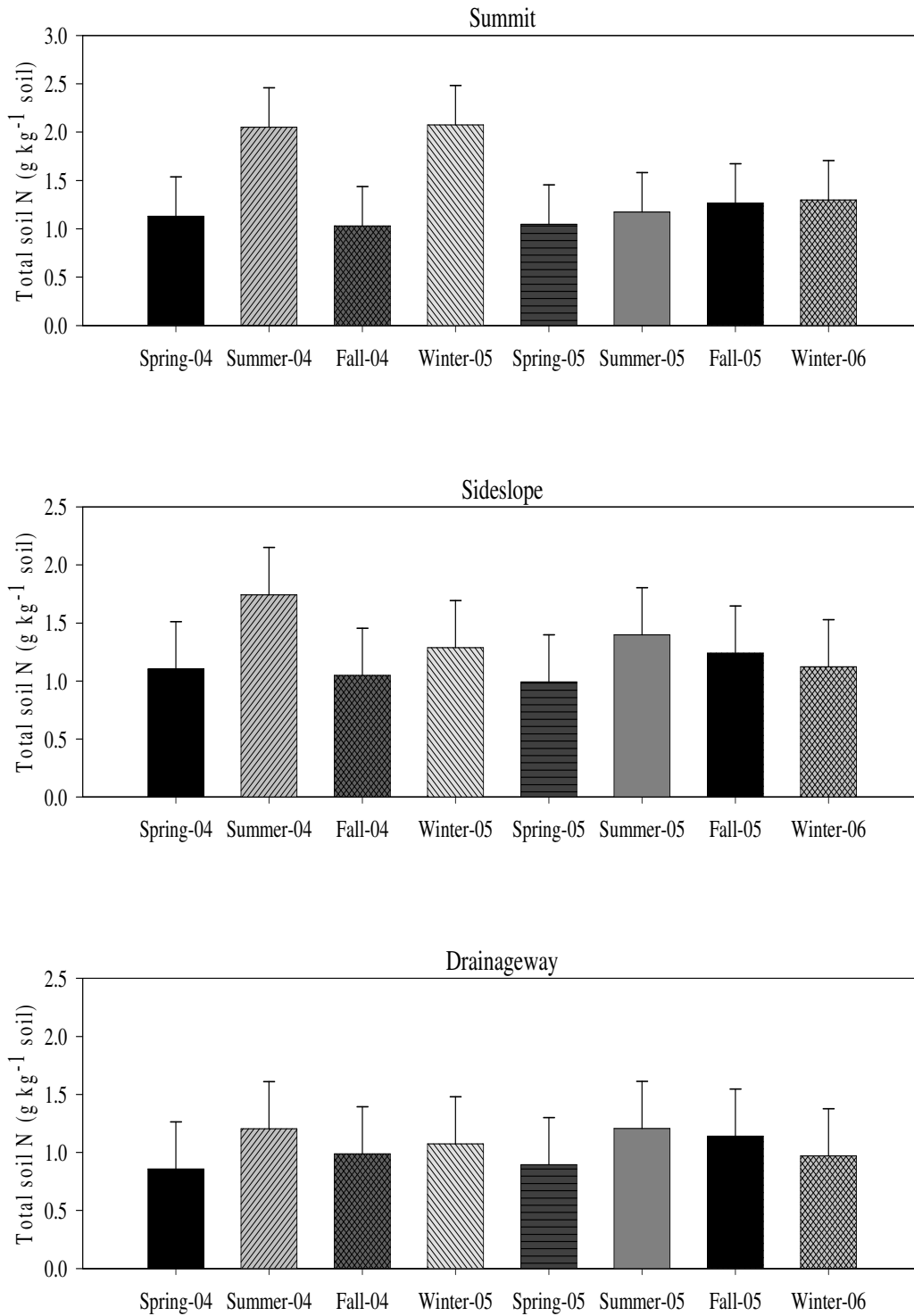


Fig. 9. Seasonal total soil N following six years of soil management. Data are averaged across soil management treatments. Bars represent standard errors of the mean.

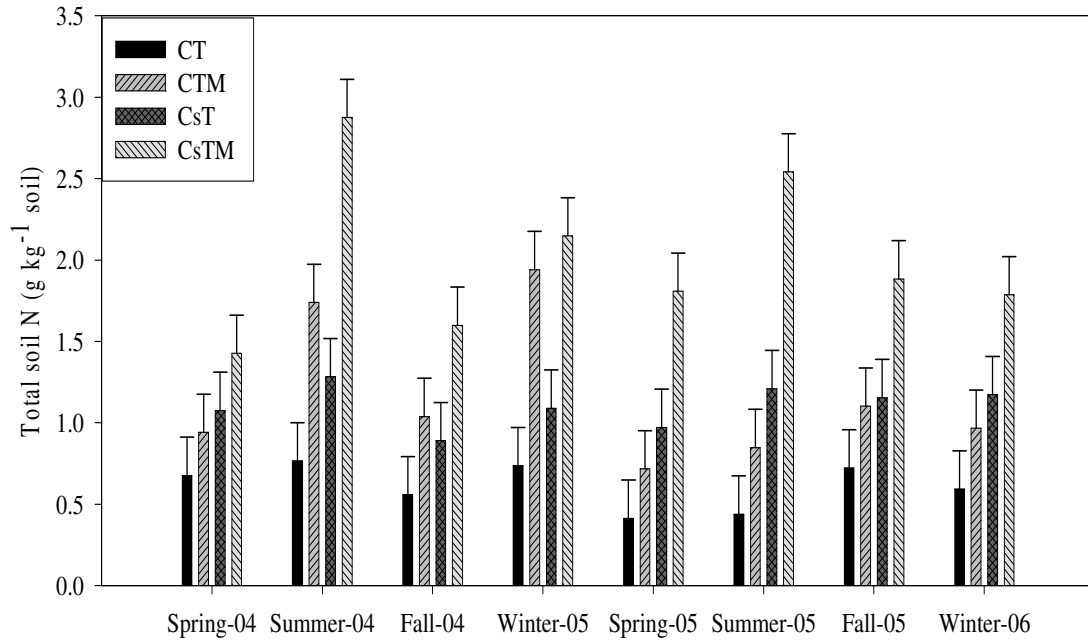


Fig. 10. Effect of soil management on total soil N averaged across summit, sideslope and drainageway landscapes. CT = Conventional tillage, CsT = Conservation tillage, CTM = Conventional tillage + dairy manure, CsTM = Conservation tillage + dairy manure. Bars represent standard errors of the mean.

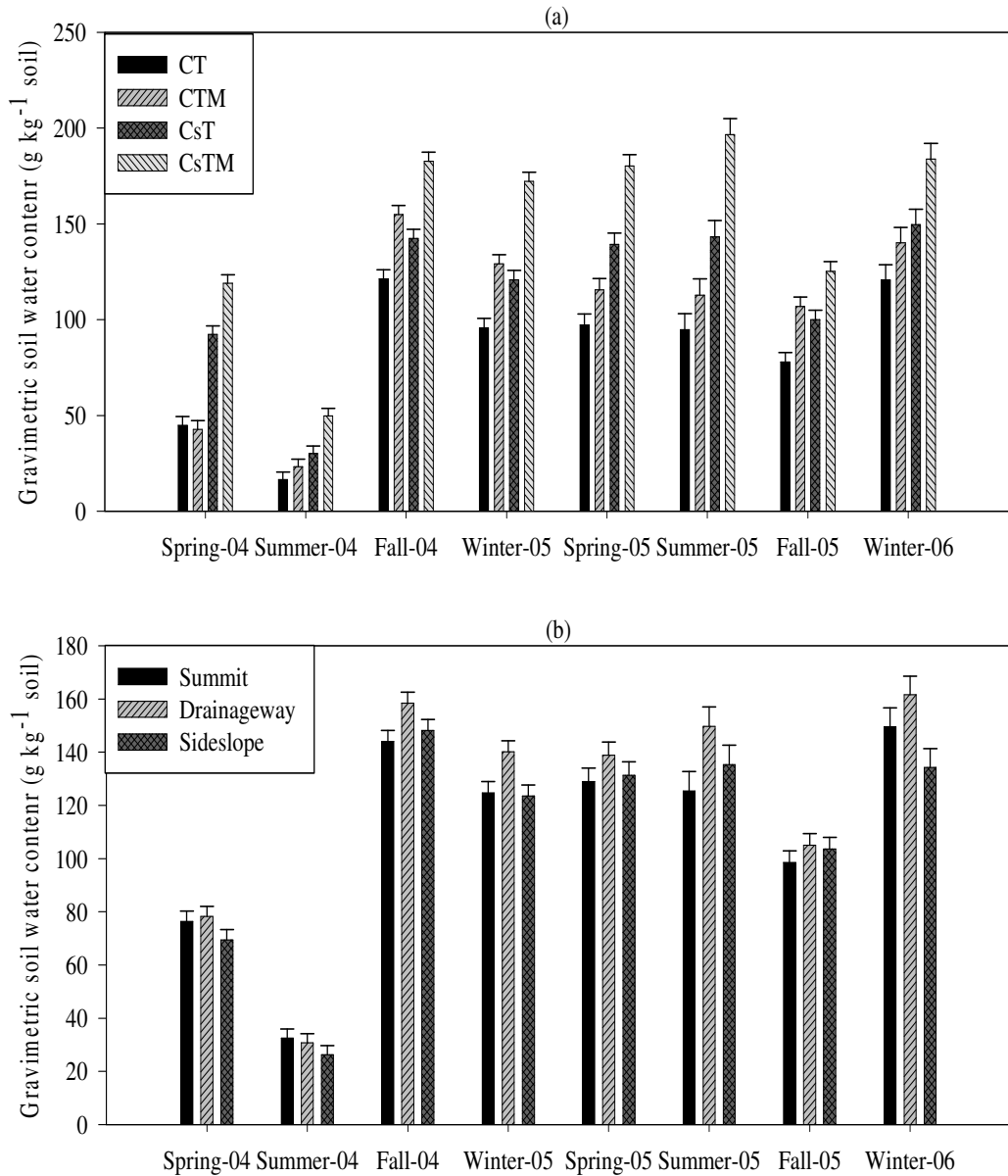


Fig. 11. Seasonal variation of gravimetric soil water content as affected by (a) soil management and (b) landscape variability. CT = Conventional tillage, CsT = Conservation tillage, CTM = Conventional tillage + dairy manure, CsTM = Conservation tillage + dairy manure. Bars represent standard errors of the mean.

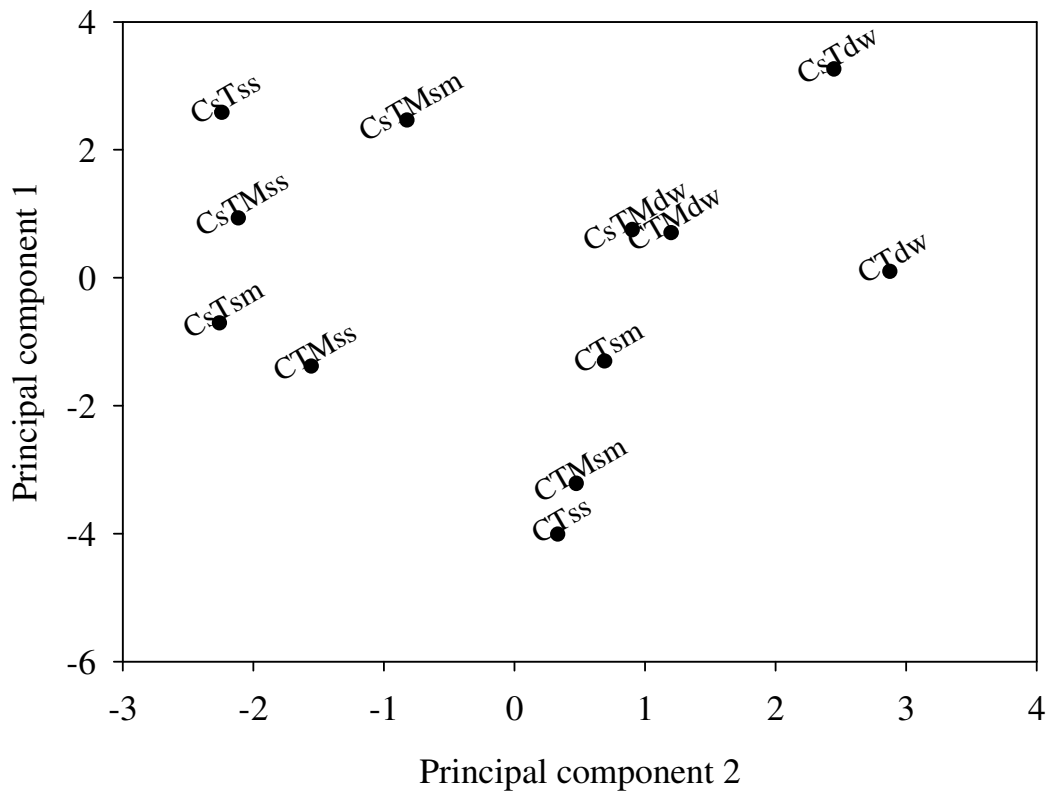


Fig. 12. Principal component scores (first two components) of normalized CH₄, N₂O and CO₂ fluxes over two years (eight seasons). The two components explain 49% of the gas flux variability. CT = Conventional tillage, CsT = Conservation tillage, CTM = Conventional tillage + dairy manure, CsTM = Conservation tillage + dairy manure. sm = summit position, ss = sideslope position, dw = drainageway position.

III. AGROECOSYSTEM MANAGEMENT EFFECTS ON SOIL CARBON AND NITROGEN MINERALIZATION ACROSS A COASTAL PLAIN CATENA

ABSTRACT

Soil management that maximizes crop production while protecting the environment requires understanding of carbon (C) and nitrogen (N) dynamics. Carbon and N dynamics are influenced by soil properties, landscape variability, agroecosystem management and climatic factors. In the southeastern US, use of cover crops in conservation tillage has increased in recent years. However, landscape scale studies evaluating C and N dynamics in relation to agroecosystem management and landscape variability are lacking. The objective of this study was to evaluate surface (0-5cm) soil C and N dynamics under conservation tillage (CsT) and conventional tillage (CT) following six years of treatment implementation. The study site is a 9-ha field containing a corn (*Zea mays* L.)-cotton (*Gossypium hirsutum* L.) rotation. Effects of fall dairy manure application on C and N mineralization in each tillage system (CsTM and CTM) were also evaluated. Conservation tillage systems included white lupin (*Lupinus albus* L.), crimson clover (*Trifolium incarnatum* L.) black oat (*Avena strigosa* Schreb.) and rye (*Secale cereale* L.) cover crops. The tillage and dairy manure treatments were located on summit, sideslope and the drainageway landscape positions on a 9-ha field at E.V. Smith Research Center, near Shorter, AL. Soil was incubated in the dark at 25°C for 182 days at 85% of field capacity. Soil mineral N concentration and CO₂-C evolution were

determined at 1, 3, 7, 14, 28, 59, 112 and 182 days. Conservation tillage had 125% higher organic C concentration than the CT treatment. Dairy manure increased soil organic C concentration by 70% and 81% on CT and CsT treatments, respectively. Carbon mineralization was in the order CsTM > CsT > CTM > CT and was similar on the three landscape positions. Total soil N showed patterns similar to organic C. Higher N mineralization was observed with CsT and CsTM treatments than on CT and CTM treatments, while higher relative N mineralization was observed on CT and CsT treatments. Landscape variability evaluated in this study was not sufficient to significantly influence C, N and C:N mineralization. However, terrain attributes satisfactorily explained the variability of C, N mineralization, relative N mineralization, C turnover and C:N mineralization on CT treatments than on CsT, CTM and CsTM treatments. It appears that on this Coastal Plain site the dynamic soil C and N properties are influenced more by management than by landscape variability.

INTRODUCTION

The southeastern US is characterized by warm, humid, conditions that favor soil organic matter decomposition and rapid loss of soil C and N mineralization (Franzluebbers, 2005). This contributes to climate change by increasing CO₂ concentration in the atmosphere. It can also contribute to groundwater contamination through NO₃ leaching. This necessitates use of environmentally friendly soil management techniques. Such techniques include conservation tillage systems that include cover crops. Cover crops are incorporated into the soil or are chemically

terminated and left on the soil surface prior to row crop planting, and contribute to soil organic matter (SOM) accumulation and general improvements in soil quality.

Soil microbial respiration is influenced by environmental factors, crop residue chemical composition, and soil management practices. Reduced tillage systems have been observed to reduce CO₂ emissions from soil. Due to minimal soil mixing, these systems emit less CO₂ compared to conventional tillage systems. Carbon dioxide emissions increase immediately following tillage operations (Calderon and Jackson, 2002; Reicosky and Archer, 2007). Amount of CO₂ released depends on the level of soil disturbance. Reicosky (1997) found that use of intensive cultivation equipment resulted in higher CO₂ emission compared to use of reduced tillage cultivation tools. Higher levels of CO₂ were released with deep cultivation compared to shallow cultivation. A review by West and Marland (2002) revealed that production of corn (*Zea mays* L.), soybean (*Glycine max* (L) Merr.) and wheat (*Tritium aestivum* L.) in the US under no-till and conventional till emits on average 137 and 168 kg CO₂-C ha⁻¹ yr⁻¹ respectively. These C flux estimates include all factors of fertilizer manufacturing, transportation and application. These data suggest that on average, a change from conventional tillage to no-till would result in reduction in amounts of C released into the atmosphere, but results will vary with location and site-specific farm operations.

Soil management strategies and inherent soil properties influence soil microbial processes, and hence, soil C and N mineralization. These mineralization processes are studied through determination of soil CO₂ evolution and mineral N at various time intervals, typically in laboratory incubation studies. Under laboratory conditions, field moist or air-dried soil that is re-wetted to attain desired moisture content is incubated at

controlled temperatures and moisture. Studies show no differences in C mineralization between mineralization of field moist soil compared to soils that are dried and re-wetted (Haney et al., 2004), except for an initial short lived C mineralization flush as observed by Franzluebbers (1999).

Landscape variability can influence soil mineralization at the field scale. Lee et al. (2006) found that field scale variability resulted in highly variable soil properties such that they could not detect significant differences in N₂O, CO₂ and CH₄ emission between conventional tillage and no-till treatments. Further, they established that emission of the three gases was more related to microbial activity and labile C and N sources than texture and other soil properties.

In the southeastern US, the use of conservation tillage and cover crops has increased in recent years. Soil organic matter decomposition is rapid due to high rainfall and temperatures in the region. Landscape-scale studies on soil C and N dynamics in relation to agroecosystem management and landscape variability in the southeastern Coastal Plain is lacking. The objective of this study was to compare soil C and N mineralization under conservation tillage and conventional tillage, with or without dairy manure application, across a Coastal Plain catena.

MATERIALS AND METHODS

Study Site

The study site is at the E.V. Smith Research Center near Shorter, AL, US, and lies at 85°53'50''W and 32°25'22''N. The site has a gentle slope ranging from 0-5%, and the soils are classified as Typic, Oxyaquic, and Aquic Paleudults. Details of the surface soil

chemical characteristics prior to experiment establishment (2000) at the site have been described by Terra et al. (2006).

Soil Management and Experimental Design

The study site is a 9-ha field containing a corn (*Zea mays* L.)-cotton (*Gossypium hirsutum* L.) rotation. Soil management treatments were established in 6.1 m wide by ~240 m long strips across the landscape (Fig. 13) in a randomized complete block design with six replications. Plots measuring 6.1 m x 18.3 m were delineated in each strip, resulting in a total of 496 plots. Soil management treatments implemented in fall 2000 include: 1) conventional tillage (CT) involving disking, chisel plowing, field cultivation (to level seedbed), 2) conventional tillage + dairy manure (CTM) applied each fall at a rate of ~ 10 Mg ha⁻¹ (fresh weight basis), 3) conservation tillage (CsT) consisting of non-inversion in-row subsoiling and winter cover crops of a legume mixture prior to corn and rye/black oat mixture prior to cotton and 4) conservation tillage + dairy manure (CsTM) applied in the fall at a rate of ~ 10 Mg ha⁻¹. Further details on experiment treatments can be found in Terra et al. (2006).

The field was divided into three soil landscape positions (Fig. 13) using an order 1 soil survey (1:5000) and a high resolution digital elevation model (DEM). Digital elevation data were obtained using a Real Time Kinematic (RTK)-GPS. Elevation data were interpolated to provide a DEM in Arc Info (ESRI, Redlands, CA), and used to develop slope and the compound topographic index (CTI) (Moore et al. 1993). The compound topographic index relates specific catchment area to slope. Soil survey data were rasterized to indicate seasonal high water table (SHWT) and overlaid with DEM, slope and CTI layers. Fuzzy k-means unsupervised clustering of these multivariate data

was used to delineate three landscape positions (summit, sideslope and drainageway) (Fridgen et al., 2004).

Thirty six GPS referenced plots were identified. Plots were distributed across the three landscape positions and four management systems cropped to cotton during 2004. These plots were under corn rotation in 2005 and under cotton in 2006. Each management treatment was replicated three times ($3 \times 4 \times 3 = 36$ plots).

Dairy manure was applied on October 22, 2004 and November 19, 2005. On CT plots, disking and plowing were done on April 29, 2004 and April 5, 2005. Dairy manure applied in fall 2004 had total P, K, Ca, and Mg of 3.4, 1.3, 29 and 8.9 g kg⁻¹ manure as determined through nitric/perchloric acid wet ashing (Hue and Evans, 1986) while total N (Kjedahl digestion) was 8.2 g kg⁻¹ manure. In fall 2005, the dairy manure had total P, K, Ca, and Mg concentration of 0.9, 0.9, 7.8 and 2.1 g kg⁻¹ manure respectively, while N content was 6.2 g kg⁻¹ manure. Dairy manure moisture content was 44% and 70% in 2004 and 2005, respectively.

Soil Collection and Preparation

Soil samples were obtained from the study site on 21 September 2006 at a depth of 0-5cm. Soil samples were obtained from the 36 GPS referenced plots on summit, sideslope and drainageway landscape positions. Soil was stored at 4°C and transported to the laboratory where it was air dried at room temperature and sieved through a 2mm sieve. Organic soil C and total soil N concentration were determined using a LECO TruSpec CN analyzer (Leco Corp., St. Joseph, MI.). Gravimetric soil moisture was determined by weighing 1 g field moist soil and drying it in the oven at 105°C to constant

weight. Following this, fresh soil weight equivalent to 50 g oven dry weight was determined and weighed for incubation.

The soil was placed in 150-mL Falcon Filter Units (micro-lysimeters) according to methods of Nadelhoffer (1990). In addition, four blank units were included to act as controls. Briefly, the Falcon Filter Unit is made up of an upper and a lower chamber. The two chambers are separated by a filter system consisting of a filter paper and glass wool. Soil was placed in the upper chamber and 100 mL of 0.01M CaCl₂ was added and allowed to equilibrate with the soil for 30 minutes. The CaCl₂ solution was leached out by applying suction at -60 kPa from a vacuum pump to remove excess moisture and any mineral N present in the soil prior to incubation. At this suction pressure, the soil attained 85% field capacity. The lysimeters (containing moist soil) were weighed to obtain baseline mass at 85 % field capacity. To maintain soil moisture at this level, lysimeters were weighed between sampling dates and deionized water was added as needed. The soil was incubated aerobically at 25°C in the dark, and the leaching procedure was repeated at 1, 3, 7, 14, 28, 59, 112 and 182 days.

The leached CaCl₂ solution was analyzed for NH₄ and NO₃ concentration colorimetrically using a μ Quant™ micro-plate spectrophotometer (BioTek instruments, Inc. Winooski, VT). The procedure involves color development by combining the CaCl₂ extract with citrate, salicylate-nitroprusside and hypochlorite reagent in micro-plate wells for NH₄ determination. Soil NO₃ concentration is measured by converting the NO₃ into NH₄ by adding Devarda's alloy and sulphuric acid prior to color development. Soil organic N concentration was calculated as the difference between total and inorganic N (NH₄ and NO₃ determined at start) (Kingery et al., 1996). Cumulative N mineralization

was obtained by summing successive inorganic N at each sampling time. Relative N mineralization was calculated by dividing cumulative inorganic N at the 182 sampling date by initial organic N concentration.

Following each leaching (except initial leaching at start), samples of CO₂ evolved from the soil were collected. The procedure involved pumping CO₂ free air at a rate of 1.5 L min⁻¹ through the soil for at least three minutes, while keeping the incubation unit valves open. All valves were then closed and the micro-lysimeters placed on the laboratory bench at room temperature for three hours to accumulate CO₂ from soil respiration. A 3 mL gas sample was drawn from the upper chamber port with a syringe and needle and transferred into a 3 mL storage vial and stored at 4°C pending CO₂ concentration determination. Carbon dioxide accumulation time was recorded to the nearest minute. Carbon dioxide concentration was determined using a Varian Star cx gas chromatograph (Varian, Walnut Creek, CA) with a 4 m Haysep R column and a ⁶³Ni electron capture detector (ECD). The detector temperature was 350°C with N₂ carrier gas at a 17 mL min⁻¹ flow rate. Percent CO₂ evolved at each sampling time (obtained from GC analysis) was converted to volume of CO₂ in each lysimeter. The volume was further converted to mass of CO₂-C per unit soil mass using the gas law. The cumulative amount of CO₂-C evolved was calculated by interpolation based on the measured CO₂-C evolution rate at each sampling time. Carbon:N mineralization was calculated by dividing cumulative CO₂-C mineralization at 182 sampling date by cumulative N mineralization at the same sampling date.

Data Analysis

The generalized linear model (GLM) in SAS (SAS Inst., Cary, NC) was used to compare terrain attributes across landscape positions and to test for effect of soil management and landscape variability on cumulative C and N mineralized. Treatment means were compared using Fisher's protected least significant difference (LSD) at $P \leq 0.05$.

Stepwise regression and correlation were used to relate landscape variability factors to C and N mineralization for each soil management treatment. Terrain attributes used in the stepwise regression and correlation analysis include CTI, digital elevation, slope, planimetric curvature, profile curvature, flow accumulation, SHWT and surface horizon sand, silt and clay content as determined by Terra et al. (2006).

Organic C, total N, cumulative C and N mineralization, relative N mineralization and C turnover data were normalized followed by principal component analysis in SAS. A plot of scores of the first two principal components was done to determine if there were distinct groupings relating to landscape and soil management treatments.

RESULTS

Landscape Variability

There were significant differences in landscape variability factors between landscape positions, except planimetric curvature and surface horizon silt content (Table 5). The sideslope landscape position had higher slope, surface horizon clay content and profile curvature compared to the drainageway position. Higher slope on the sideslope result in runoff that would accumulate in the drainageway. This is depicted by higher

flow accumulation, surface horizon sand content and compound topographic index (CTI) within the drainageway landscape. Positive profile curvature values found on the summit landscape position indicate a convex profile, while negative profile curvatures on drainageway indicate concave profiles (Li et al., 2005). Highest elevations and depth to seasonal high water table (SHWT) were found on the summit.

Soil Carbon Mineralization

Total soil organic C concentration was affected by tillage and dairy manure treatments ($P = 0.001$) following six years of tillage and dairy manure application (Table 6). Total soil organic C concentration was similar across the three landscape positions. There were no interactions between landscape position and soil management treatments (CT, CTM, CsT and CsTM) on total soil organic C concentration. Averaged across landscape positions, total organic C concentration was in the order $CsTM > CsT > CTM > CT$. Dairy manure increased soil organic C by 70% and 81% on CT and CsT treatments, respectively (Table 6). On both CT and CsT treatments, dairy manure application approximately doubled total soil organic C concentration on the summit and in the drainageway. Positive correlation was observed between soil organic C and surface horizon sand content on the four treatments, while surface horizon clay content had negative correlation with soil organic C (Appendix 3). However, these correlations were not significant at $P \leq 0.05$.

Cumulative C mineralization (averaged across landscape positions) was highest on CsTM treatments ($1146 \text{ mg kg}^{-1} \text{ soil}$) and lowest on CT treatments (417 mg kg^{-1}) (Fig. 14). Conservation tillage treatments mineralized 28% more soil C than CT treatments,

while CsTM mineralized 48% more soil C than CsT treatments (Fig. 14). Dairy manure increased soil C mineralization in both CT and CsT treatments although the increase was significant only on CsT tillage systems. Carbon mineralization was similar ($P = 0.618$) across landscapes. Though not significantly different ($P = 0.592$), C turnover was higher on CT and CsT treatments compared to CTM and CsTM treatments (Table 7).

Surface horizon silt content accounted for at least 70% of the variation in cumulative C mineralized in CT (Table 8), CTM (Table 9) and CsT (Table 10) treatments. Surface horizon soil clay content had a negative effect on C mineralization in the four soil management treatments (Tables 8, 9, 10 and 11). Overall, C turnover was low, ranging between 6.3% (CT) and 4.2% (CsTM) (Table 7). When the four soil management treatments were pooled, C turnover and C:N mineralized were positively correlated with surface horizon soil sand content, and negatively correlated with surface horizon soil clay content (data not shown). Neither soil management ($P = 0.820$) nor landscape position ($P = 0.502$) had a significant effect on C:N mineralized (Table 7). Also, terrain attributes had no significant influence on C:N mineralized on CT (Table 8) and CsT (Table 10) treatments.

Soil N Mineralization

Total soil N concentration after six years of tillage and dairy manure treatments showed patterns similar to those of organic C concentration. Conservation tillage and CsTM treatments had higher ($P = 0.001$) total soil N concentration (Table 6) than CT and CTM treatments, with no differences between landscape positions ($P = 0.364$). There were no significant interactions between landscape position and soil management

treatments ($P = 0.590$) on total soil N concentration. No significant correlations were observed between total N concentration and terrain attributes (Appendix 3). Also, no terrain attribute contributed significantly to total N variability on CT (Table 8) treatments. Profile curvature had a negative influence on soil total N on CsT treatments, while it had a positive effect on CsTM treatments (Tables 10 and 11). Profile curvature explained 52% and 38% of total N variability on CsT and CsTM treatments, respectively.

Cumulative soil N mineralization was highly influenced ($P = 0.001$) by soil management treatments, but was unaffected by landscape position ($P = 0.244$) (Appendix 4). There were no significant interactions between landscape position and soil management ($P = 0.872$) on cumulative soil N mineralized. Highest N mineralization was observed on CsTM treatment, while the lowest N mineralization was observed on CT treatments (Fig. 15) across all three landscape positions. Averaged across landscape positions, CsT treatments mineralized higher cumulative N than CT treatments. Dairy manure had no effect on cumulative N mineralization on CT systems, while it increased N mineralization on CsT systems by about 17% (Fig. 15). The ratio C:N mineralized (Table 7) was similar in the four soil management treatments and on the three landscape positions. None of the terrain attributes contributed significantly to N mineralization variability on CsT and CsTM treatments (Tables 10 and 11). On CT and CTM treatments, the surface horizon soil clay content explained at least 30% of N mineralization variance (Tables 8 and 9). An increase in soil clay content corresponded with an increase in N mineralization.

Higher relative N mineralization ($P = 0.001$) was observed on CsT and CT treatments compared to CsTM and CTM treatments (Table 7). Landscape position had

no effect ($P = 0.124$) on relative N mineralization, but higher relative mineralization occurred on CsT and CT treatments compared to CsTM and CTM treatments (Table 7). Over 40% of relative N mineralization variance was explained by surface horizon silt content on CT treatments (Table 8), while no landscape variability factor significantly explained relative N mineralization on CsT treatments (Table 10).

Carbon and N Mineralization Multivariate Analysis

Principal component analysis assigns eigenvalues to each principal component ranked by its contribution to measured data variability. The first two principal components explained 81% of C and N mineralization variance. A plot of scores of the first two principal components (Fig. 16) shows mineralization data can be categorized into two groups based on tillage. No distinction can be made on a landscape position basis, but stepwise regression suggested certain terrain attributes significantly influenced C and N mineralization. These effects are discussed under each dependent variable below.

DISCUSSION

Soil Carbon Mineralization

Soil C accumulation and loss through decomposition and mineralization are impacted by soil management and soil microclimate resulting from spatial landscape variability. In this study, landscape position had no effect on soil organic C after six years of tillage and dairy manure application (Table 6). Consequently, landscape position had no effect on soil C mineralization. Higher soil organic C concentration on CsT and

CsTM treatments was due to external inputs of SOM through cover crop residues and dairy manure application. Higher soil organic C concentration on CsT compared to CT treatments may also be attributed to lower soil disturbance on CsT treatments and lower SOM decomposition rate. Similarly, Hussain et al. (1999) found higher organic C concentration on no-till than on moldboard and chisel plowed treatments on surface soil in Illinois (US), after eight years of tillage. Carbon mineralization decreased with increasing surface horizon clay content in our study. This is similar to findings of Franzluebbers (1999), who found a decrease in relative C mineralization with increase in soil clay content that may be due to physical protection of organic matter by clay.

Although CsT increased soil organic C concentration in the surface soil, decline in organic C concentration in the subsoil may occur due to lack of incorporation of residues under these systems (Ai-Zhen et al., 2007). Higher soil C mineralization on CsT and CsTM treatments compared to CT and CTM treatments and may be attributed to higher soil organic C on these treatments (Table 6). The high organic C concentration on CsT and CsTM treatments and higher C mineralization on the same treatments suggest that these treatments enhanced soil C buildup that was readily mineralized under laboratory conditions. Similarly, Oorts et al. (2006) found higher C and N mineralization from no-till soils compared to conventionally tilled soils. They associated higher mineralization on no-till soils with higher C and N in the particulate organic matter on these treatments. Higher C turnover on CT and CsT compared to CTM and CsTM treatments in our study were due to lower organic C concentration in those treatments (Table 6). Higher C turnover on CT and CsT treatments may lead to faster depletion of soil organic C in these treatments compared to CTM and CsTM.

Soil Nitrogen Mineralization

After six years of tillage, CsT tillage had higher total soil N compared to CT treatment. Conventional tillage causes soil disturbance and results in higher SOM loss through decomposition processes, while CsT conserves SOM. Dairy manure increased total N concentration on both CT and CsT treatments, and provided substrate for microbial respiration. This resulted in higher N mineralization on CsTM and CTM treatments (averaged across landscape positions) compared to CT and CsT treatments (Fig. 15). Higher total soil N in CsTM and CTM treatments corresponded with lower relative N mineralization compared to CT and CsT treatments, indicating buildup of total soil N in the dairy manure treatments. Dairy manure application resulted in net SOM buildup in both tillage systems, despite the higher C and N mineralization observed on these treatments compared to no dairy manure treatments. On the contrary, higher relative N mineralization on CT and CsT treatments is indicative of faster depletion of soil organic N on these treatments compared to CsTM and CTM treatments. This suggests that CT and CsT treatments would require higher inorganic N inputs than CTM and CsTM treatments for crop production in this environment.

Similar C:N mineralization in the four soil management treatments is indicative of similar SOM quality among the treatments. The same can be said of SOM quality across the three landscape positions. Although there were significant differences in terrain attributes between landscape positions (Table 5), these did not translate into differences in N mineralization between landscapes. Gilliam et al. (2005) found no differences between *in situ* and laboratory N mineralization rates of soil along a watershed gradient in West Virginia, US. They concluded the difference in N mineralization between

watersheds was due to dynamic soil characteristics (microbial communities and soil chemical composition) rather than physical differences (elevation, aspect and slope) among sites.

Stepwise regression indicates that a relatively higher number of terrain attributes were related to N mineralization, C:N mineralization and relative N mineralization variance on CT treatments than on CTM, CsT and CsTM treatments (Tables 8, 9, 10, and 11). These terrain attributes also affect soil moisture, and their influence on N mineralization variance is most likely related to this. It is important to note that the influence of these terrain attributes on C and N dynamics may not translate to laboratory measurements. Increase in N mineralization with increase in surface horizon clay content may be related to C mineralization. Decrease in C mineralization results in reduced N immobilization and favors net N mineralization as observed by Franzluebbers (1999).

Carbon and N Mineralization Principal Component Analysis

Most (81%) C and N mineralization variability was explained by the first two principal components. Carbon, N, C:N, relative N mineralization and C turn over can be grouped into two clusters (Fig. 16). The clusters are dependent on tillage system irrespective of landscape position. This suggests that although terrain attributes influenced C and N dynamics, tillage played a greater role.

CONCLUSIONS

Despite the high SOM decomposition rates experienced in the southeastern US, six years of CsT and dairy manure application increased soil organic C and total N concentration in the soil. Dairy manure increased total organic C and total soil N on both CT and CsT systems, with higher increases observed on CsT treatments. Higher soil C and N mineralization on CsT and CsTM treatments was due to contribution of SOM from dairy manure and cover crop residues, and perhaps increases in soil microbial populations and diversity.

Landscape variability was not sufficient to significantly influence C respiration and N mineralization. A higher number of terrain attributes contributed to C and N mineralization variance on CT treatments than on CTM, CsT and CsTM treatments. This may be related to relative enhancement of soil moisture (and related soil microclimate) by various terrain attributes on CT treatments.

Although terrain attributes influenced C and N dynamics, tillage had greater impact on these dynamics as shown by the clusters formed from principal component scores (Fig. 16). Dairy manure treatments (CTM and CsTM) had higher scores than CT and CsT treatments.

It is apparent that conservation tillage and dairy manure application can increase soil organic C and N concentration while contributing to inorganic N mineralization for crop uptake in the southeastern US Coastal Plains.

Table 5. Analysis of variance of terrain attributes among summit, drainageway and sideslope landscape positions at E.V. Smith Research Center near Shorter, AL.

Terrain attribute	Soil landscape position†			P-value
	Summit	Sideslope	Drainageway	
CTI‡	3.98b	4.15b	6.16a	0.0001
Elevation (m)	71.33a	70.53b	69.49c	0.0001
Planimetric curvature	0.01a	-0.01ab	-0.08b	0.0620
Profile curvature	0.02a	0.02a	-0.09b	0.0030
Slope (%)	0.60c	3.33a	1.33b	0.0001
Flow accumulation	05.01b	7.13b	30.35a	0.0040
SHWT (cm)§	145.83a	108.33b	75.00c	0.0010
Sand (%)¶	56.78b	54.32b	63.75a	0.0001
Silt (%)	24.44a	25.50a	25.20a	0.7360
Clay (%)	18.79a	21.12a	11.06b	0.0001

† Values followed by the same letter within rows are not significantly different at $P \leq 0.05$.

‡ Compound topographic index

§ Seasonal high water table

¶ Surface horizon sand, silt and clay content

Table 6. Soil organic C and total N affected by six years of tillage and dairy manure application at E.V. Smith Research Center near Shorter, AL. Soil management treatments were across three landscape positions

Soil management treatment [†]	Soil organic C	Soil total N
	g kg ⁻¹ soil	
CsTM	27.2	2.1
CsT	15.0	1.3
CTM	11.3	1.1
CT	6.7	0.8
LSD [‡]	2.0	0.1
ANOVA		
Source of variation	P-value	
Soil management (M)	0.0001	0.0001
Landscape (L)	0.1900	0.3644
M x L	0.4820	0.5904

[†] CT, conventional tillage; CTM, conventional tillage with dairy manure, CsT, conservation tillage; CsTM, conservation tillage with dairy manure.

[‡] Least significant difference at $P \leq 0.05$

Table 7. Soil C turnover, relative N, and C:N mineralized affected by six years of tillage and dairy manure application at E.V. Smith Research Center near Shorter, AL. Soil management treatments were across three landscape positions

Soil management treatments [†]	C turnover	Relative N mineralization	C:N mineralized
	—————%—————		
CsT	5.0	5.9	10.0
CT	6.3	4.8	12.4
CsTM	4.2	4.4	13.2
CTM	4.7	3.1	13.4
LSD [‡]	NS [§]	0.9	NS
ANOVA			
Source of variation	P-value		
Soil management (M)	0.592	0.001	0.820
Landscape (L)	0.484	0.124	0.502
M x L	0.793	0.707	0.640

[†]CT, conventional tillage; CTM, conventional tillage with dairy manure; CsT, conservation tillage; CsTM, conservation tillage with dairy manure.

[‡] Least significant difference

[§] Not significant at $P \leq 0.05$

Table 8. Stepwise regression relating landscape variability factors to organic C, total N, relative N mineralization, C turnover and C:N mineralized on conventional tillage (CT) treatments.

Dependent variable	Independent variable†	Partial R ²	P-value
Cumulative C mineralized	Silt (-)‡	0.984	0.0001
	Clay (-)	0.016	0.0001
Cumulative N mineralized	Silt (+)§	0.476	0.0400
	Clay (+)	0.291	0.0340
Total N	None	-	-
Organic C	Elevation (+)	0.364	0.0850
C:N mineralized	None	-	-
Relative N mineralized	Silt (+)	0.417	0.0600
	SHWT (-)‡	0.198	0.1300
	Clay (+)	0.153	0.1300
	Planimetric curvature (+)	0.150	0.0550
Carbon turnover	None		

† Silt and clay = surface horizon silt and clay contents. SHWT = seasonal high water table.

‡ An increase in the independent variable results in a decrease in dependent variable in the regression model

§ An increase in independent variable results in an increase in the dependent variable in the regression model

Table 9. Stepwise regression relating landscape variability factors to organic C, total N, relative N mineralization, C turnover and C:N mineralized on conventional tillage + dairy manure (CTM) treatments.

Dependent variable	Independent variable†	Partial R ²	P-value
Cumulative C mineralized	Silt (-)‡	0.986	0.0001
	Clay (-)	0.031	0.0001
Cumulative N mineralized	SHWT (+)§	0.402	0.0670
	Clay (+)	0.386	0.0160
	Planimetric curvature (-)	0.079	0.1450
	Flow accumulation (-)	0.124	0.0020
Total N	Planimetric curvature (+)	0.465	0.0430
Organic C	None	-	-
C:N mineralized	SHWT (+)	0.409	0.0640
	Planimetric curvature (-)	0.329	0.0340
Relative N mineralized	Clay (+)	0.412	0.0630
Carbon turnover	Planimetric curvature (+)	0.470	0.0410

† Silt and clay = surface horizon silt and clay contents. SHWT = seasonal high water table.

‡ An increase in the independent variable results in a decrease in dependent variable in the regression model

§ An increase in independent variable results in an increase decrease in the dependent variable in the regression model

Table 10. Stepwise regression relating landscape variability factors to organic C, total N, relative N mineralization, C turnover and C:N mineralized on conservation tillage (CsT) treatments.

Dependent variable	Independent variable†	Partial R ²	P-value
Cumulative C mineralized	Silt (-)‡	0.712	0.0040
	Clay (-)	0.288	0.0001
Cumulative N mineralized	None	-	-
Total N	Profile curvature (-)	0.521	0.0280
Organic C	None	-	-
C:N mineralization	None	-	-
Relative N mineralized	None	-	-
C turnover	profile curvature (-)	0.550	0.0220

† Silt sand and clay = surface horizon silt and clay contents

‡ An increase in the independent variable results in a decrease in dependent variable in the regression model

Table 11. Stepwise regression relating landscape variability factors to organic C, total N, relative N mineralization, C turnover and C:N mineralized on conservation tillage + dairy manure (CsTM) treatments.

Dependent variable	Independent variable†	Partial R ²	P-value
Cumulative C mineralized	Elevation (+)‡	0.698	0.0050
	SHWT (+)	0.171	0.0310
	Silt (-)§	0.062	0.0870
	Clay (-)	0.069	0.0001
Cumulative N mineralized	None	-	-
Total N	Planimetric curvature (+)	0.375	0.0800
Total C	Slope (-)	0.357	0.0890
C:N mineralized	Slope (-)	0.393	0.0710
Relative N mineralized	slope (+)	0.301	0.1260
C turnover	Planimetric curvature (+)	0.279	0.1440

† Silt and clay = surface horizon silt and clay contents. SHWT = seasonal high water table.

‡ An increase in independent variable results in an increase in the dependent variable in the regression model

§ An increase in the independent variable results in a decrease in dependent variable in the regression model

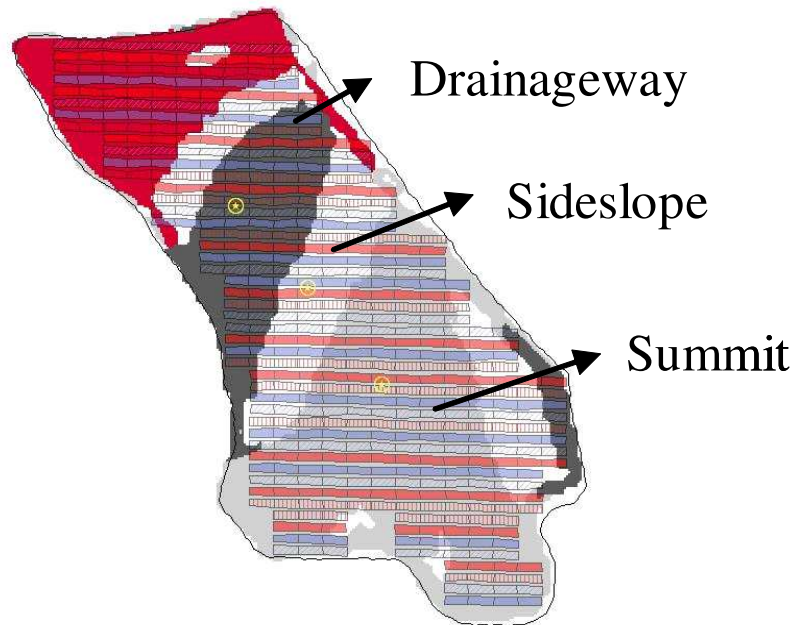


Fig. 13. Study site soil landscape positions created using fuzzy k-means unsupervised clustering based on seasonal high water table, digital elevation, slope and compound topographic index. Summit is the highest position, drainageway the lowest position, while sideslope is an eroded landscape.

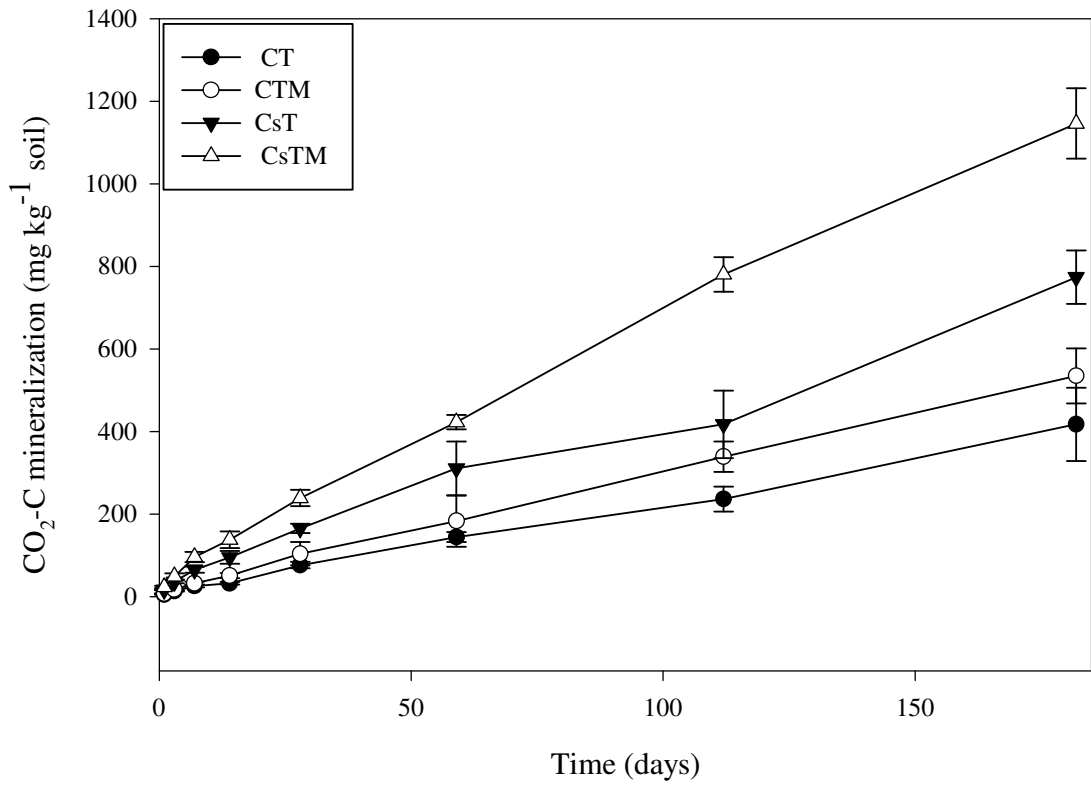


Fig. 14. Cumulative C mineralization of soil subjected to six years of tillage and dairy manure application. Data is mean of three landscape positions. CT = Conventional tillage, CsT = Conservation tillage, CTM = Conventional tillage + dairy manure, CsTM = Conservation tillage + dairy manure. Bars are standard errors of the mean.

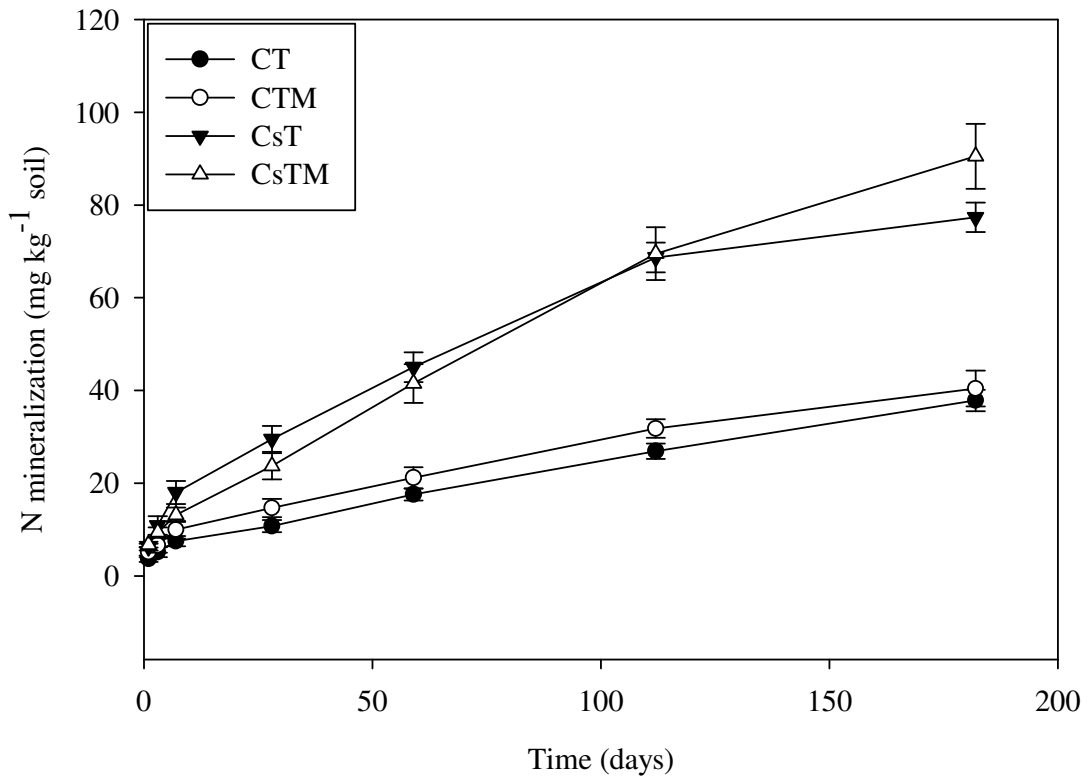


Fig. 15. Cumulative N mineralization of soil following six years of tillage and dairy manure application at E.V. Smith Research Center near Shorter, AL. Data is mean of three landscape positions. CT = Conventional tillage, CsT = Conservation tillage, CTM = Conventional tillage + dairy manure, CsTM = Conservation tillage + dairy manure. Bars are standard errors of the mean.

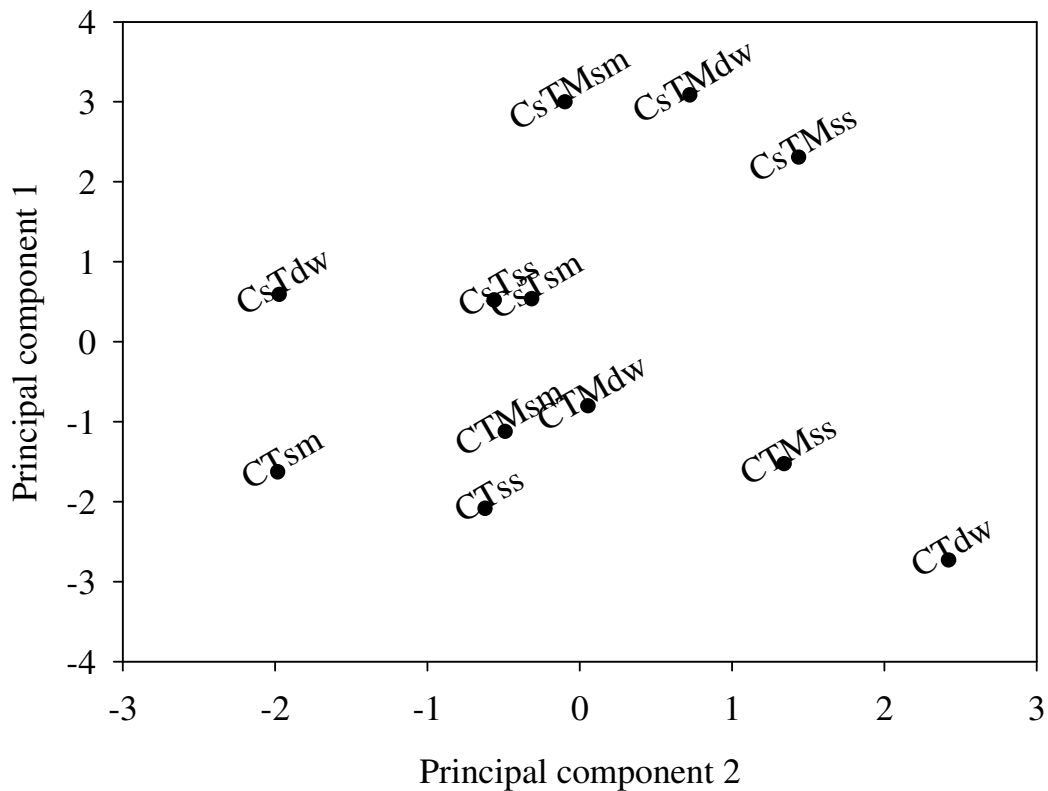


Fig. 16. Carbon and N mineralization principal component scores of the first two principal components. Data includes organic C, total N, cumulative C and N mineralization, relative N mineralization, C turnover and C:N mineralization. CT = Conventional tillage, CsT = Conservation tillage, CTM = Conventional tillage + dairy manure, CsTM = Conservation tillage + dairy manure. sm = summit position, ss = sideslope position, dw = drainageway position.

IV. AGROECOSYSTEM MANAGEMENT EFFECTS ON COVER CROP
DECOMPOSITION AND MINERALIZATION ACROSS A COASTAL PLAIN
CATENA

ABSTRACT

Cover crops improve soil properties and provide nutrients that become available to subsequent crops following decomposition and mineralization. In order to maximize nutrient availability to succeeding crops, an understanding of decomposition and nutrient release patterns of crop residues is required. This is particularly important in the southeastern US where conservation tillage with the inclusion of cover crops has increased in the recent years. Cover crop decomposition is influenced by soil properties arising from landscape variability among other factors. However, studies on cover crop decomposition at the landscape-scale in this region are lacking. Mineralization of black oat (*Avena strigosa* Schreb.) cv. SoilSaver, crimson clover (*Trifolium incarnatum* L.) cv. AURobin, spring forage rape (*Brassica napus* L.) cv. Liform and white lupin (*Lupinus albus* L.) cv. AUHomer, amended soil were studied under laboratory and field conditions. In the laboratory after 30 days, potential N and C mineralization were similar for crimson clover, spring forage rape and white lupin amended soil; however black oat amended soil showed net N immobilization. In the field, decomposition and mineralization of the cover crop residues were studied using nylon litter bags over a six month period. The

litter bags were buried or surface applied on summit, sideslope and drainageway landscape positions. Mass, C and nutrients remaining in the residues were determined at 1, 2, 4, 6, 10, 14, 18, 22 and 26 weeks. Due to similarities in initial chemical parameters of the cover crops, no significant treatment differences were observed in decomposition rate constants (k) of crimson clover, spring forage rape and white lupin. However, black oat had a lower k related to its relatively higher initial neutral detergent fiber (NDF). Buried residues lost mass and released nutrients faster than those that were surface applied, and had mean mass loss k of -0.07 and -0.017 day^{-1} respectively. All surface applied materials (except black oat) immobilized nitrogen (N) for at least 14 days, after which net N mineralization occurred. Constant residue decomposition rates were observed after 65-70 and 170 days on buried and surface placed materials, respectively. No significant differences were observed in crop residue mass, C, N, and K loss among the three landscape positions. Overall, cover crop decomposition and mineralization were influenced by residue chemical composition and placement method.

INTRODUCTION

Decline in soil fertility on continuously cultivated land, and concern for environmental protection necessitates use of soil management that sustains crop production while protecting the environment. Continuous cultivation promotes breakdown of soil organic matter (SOM) and exposes soil to erosion, thus degrading soil quality and contaminating surface water. Breakdown of SOM increases carbon dioxide (CO_2) concentration in the atmosphere, and promotes climate change. Conservation

tillage can reduce SOM decomposition rates and possibly reduce CO₂ emission into the atmosphere (Al-Kaisi and Yin, 2005).

Potential C and N mineralization gives an indication of soil biological activity. This is important in soils receiving organic inputs such as crop residues and animal manures that contribute significant amounts of inorganic N upon decomposition, a process that takes place over time. Although N mineralization under laboratory conditions may differ from that under field conditions, it can be used to give an indication of N mineralization in the field. It, however, poses the challenge of not being able to account for unpredictable flushes of N release under drying and re-wetting conditions that occur in the field.

Soil C and N mineralization studies have varied in the length of incubation. Stanford and Smith (1972) used a 30 week laboratory incubation procedure that involved leaching mineral N over pre-determined time intervals. Nitrogen mineralization constants were determined to give an index of N mineralization. Collins and Allison (2002) conducted a 198-week incubation study on an N rich grassland soil in Connecticut, US, and concluded that 30 weeks proposed by Stanford and Smith (1972) was insufficient to estimate N mineralization potential at their site. Other studies have involved laboratory incubation over a 30-day period (Franzluebbers, 1999), with an initial and final soil leaching with KCl to determine soil mineral N. Haney et al. (2001) conducted a 24 hour laboratory incubation to measure C mineralization through CO₂ evolution measurements in search of a rapid method for measuring soil N mineralization. They found that C mineralization within 24 hrs after wetting the soil was closely

correlated to 24-day N mineralization on manure amended soils, and could be used as a rapid method to determine soil N mineralization in manured soils.

Cover crops provide ground cover that reduces soil erosion and conserves soil moisture. The cover crops increase water infiltration into the soil by increasing soil aggregate stability (Liu et al., 2005). They also suppress weed growth by shading the weeds (Fisk et al., 2001), and by production of allelopathic substances (Dhima et al., 2006)). When cover crops residues decompose and mineralize, they provide plant nutrients (Dinesh et al., 2003). The benefit of this practice depends on synchrony between cover crop nutrient release and peak crop nutrient demand. It also helps reduce the amount of nutrients, particularly N, that are applied as fertilizer and manure thereby reducing chances of excess nutrients getting into the environment. This synchrony is enhanced by establishing appropriate cover crop termination timing in relation to row crop planting.

Nutrient contents and other quality parameters such as lignin and polyphenol contents influence decomposition and mineralization patterns of plant materials. Schaffers et al. (1998) found that dry matter and N losses could be explained by the initial C:N ratio, while P loss was best explained by initial P concentration in hay in a six week study in Germany. But in a long term study, Kenneth et al. (1998) found that decomposition rate of senesced forest tree leaves was determined by soluble C substrates rather than by nutrient content. These different observations suggest that external factors, mainly environmental, play a role in litter decomposition. Kenneth et al. (1998) observed faster decomposition rates in summer when there was ample soil moisture and high temperatures. Generally, most plant materials decompose rapidly during initial stages of

decomposition due to the presence of easily decomposable components. Thereafter, slow decomposition of the more recalcitrant components occurs. According to Oliver et al. (2004), this slow phase is accompanied by increase in bacteria diversity but decrease in bacterial population size. High bacteria diversity is necessary for decomposing the more recalcitrant material.

Environmental factors, particularly moisture and temperature, play an important role in crop residue decomposition. These parameters can vary considerably as a result of landscape variability. Oliver et al. (1998) found higher decomposition rates at cooler, wetter elevations. Quemada and Cabrera (1997) found maximum N mineralization (28% of N applied) of crimson clover residue occurred at -0.52 MPa moisture content and 35°C temperature under laboratory conditions. Results also showed that effect of moisture on crimson clover mineralization was enhanced by an increase in temperature, with significant temperature by moisture interactions.

Conservation tillage with the inclusion of cover crops has become increasingly popular in recent years in the southeastern US. Cover crop decomposition is influenced by soil properties arising from landscape variability among other factors. Soil properties vary within short distances, with greater variation occurring on landscapes of varying topography. However, studies on commonly used cover crop decomposition at the landscape-scale in this region are lacking. The objectives of this study were: 1) to determine potential C and N mineralization of cover crop amended soil, and 2) to determine effects of tillage and landscape variability on decomposition and nutrient release of four cover crops.

MATERIALS AND METHODS

Study Site

The study site is at the E.V. Smith Research Center near Shorter, AL, US, and lies at 85°53'50''W and 32°25'22''N. The site has a gentle slope ranging from 0-5%, and the soils are classified as Typic, Oxyaquic, and Aquic Paleudults. Details of the surface soil chemical characteristics prior to experiment establishment (2000) at the site have been described by Terra et al. (2006).

Soil Management and Experimental Design

The study site is a 9-ha field containing a corn (*Zea mays* L.)-cotton (*Gossypium hirsutum* L.) rotation. Soil management treatments were established in 6.1 m wide by ~240 m long strips across the landscape (Fig. 17) in a randomized complete block design with six replications. Plots measuring 6.1 m x 18.3 m were delineated in each strip, resulting in a total of 496 plots. Soil management treatments implemented in fall 2000 include: 1) conventional tillage (CT) involving disking, chisel plowing, field cultivation (to level seedbed), 2) conventional tillage + dairy manure (CTM) applied each fall at a rate of ~ 10 Mg ha⁻¹ (fresh weight basis), 3) conservation tillage (CsT) consisting of non-inversion in-row subsoiling and winter cover crops of a legume mixture prior to corn and rye/black oat mixture prior to cotton and 4) conservation tillage + dairy manure (CsTM) applied in the fall at a rate of ~ 10 Mg ha⁻¹. Further details on experiment treatments can be found in Terra et al. (2006).

The field was divided into three soil landscape positions (Fig. 17) using an order 1 soil survey (1:5000) and a high resolution digital elevation model (DEM). Digital

elevation data were obtained using a Real Time Kinematic (RTK)-GPS. Elevation data were interpolated to provide a DEM in Arc Info (ESRI, Redlands, CA), and used to develop slope and the compound topographic index (CTI) (Moore et al. 1993). The compound topographic index relates specific catchment area to slope. Soil survey data were rasterized to indicate seasonal high water table (SHWT) and overlaid with DEM, slope and CTI layers. Fuzzy k-means unsupervised clustering of these multivariate data was used to delineate three landscape positions (summit, sideslope and drainageway) (Fridgen et al., 2004).

Thirty six GPS referenced plots were identified. The plots were distributed across the three landscape positions and four management systems cropped to corn during 2005. Each management treatment was replicated three times ($3 \times 4 \times 3 = 36$ plots). On each landscape position, one CT plot and one CsT plot (with no dairy manure application) were selected. Litter bags (containing cover crop residues) were buried on the CT plots and surface applied on CsT plots on May 12, 2005. This resulted in a total of 6 plots ($3 \times 2 = 6$). Litter bag replication (4 replications) was done within the plots. Disking and field cultivation on the CT plots was performed on April 15, 2005.

Plant Material Collection and Chemical Composition Determination

Aboveground biomass of black oat, crimson clover, spring forage rape and white lupin were hand collected at the E.V. Smith Research center on 13 March 2005. Biomass consisted of leaves and young stems of cover crops in their vegetative growth stage. The biomass was air-dried for several days in a greenhouse with occasional turning. Drying time varied according to the moisture content of the material. Dry matter content of air

dry material was determined by oven drying at 55°C for 48 hours. This was used to determine quantity of biomass equivalent to 5 Mg ha⁻¹ on oven dry basis, the rate at which materials were applied in laboratory and the field studies. Biomass subsamples were ground and passed through 1-mm sieve and were analyzed for total organic C and N using Leco-600 analyzer (Leco Corp., St. Joseph, MI). Lignin, cellulose, hemicellulose and neutral detergent fiber (NDF) contents were determined via acid-detergent fiber (Anderson and Ingram, 1989).

Cover Crop Amended Soil Potential C and N Mineralization

Soil was sampled at 0-5 cm depth on 20 November 2006 from three conventionally tilled plots on summit landscape position. Soil samples were transported to the laboratory where they were composited into one large sample, air dried at room temperature, and sieved through a 2-mm sieve. Organic soil C and total soil N were determined using LECO TruSpec CN analyzer. The soil contained 8.4 and 0.7 g kg⁻¹ soil organic C and total N, respectively. Gravimetric moisture content was determined by drying 1 g soil at 105°C. Fifty gram oven dry equivalent soil was mixed with ground (1-mm sieve) cover crop residues at a rate of 5 Mg ha⁻¹. Amount of plant material mixed with 50 g soil was determined on soil weight basis using the convention that there is about 2.2 million kg of soil ha⁻¹ to a normal plow layer of 15 cm (Brady and Weil, 1996). Based on this, 0.11 g ground crop residue was mixed with 50 g oven dry equivalent soil in a plastic cup. Crop residues used included black oat, crimson clover, spring forage rape and white lupin. Soil moisture content was adjusted to 85% of field capacity by adding deionized water.

The plastic cups containing cover crop amended soil were placed in 1-L Mason jars containing 20 mL deionized water to maintain constant humidity. Eight mL of 1N NaOH solution (in a vial) was placed in each jar to absorb CO₂ evolved from soil microbial respiration. Jars were fitted with air-tight lids and incubated at 25°C in the dark for 30 days. Blank jars (without soil), prepared in the same way as other jars, were included as controls to determine atmospheric CO₂. A soil only (control) was included to determine un-amended mineralization. The experiment was arranged in a completely randomized design with six treatments (four cover crop residues, one control soil and one blank) replicated three times. Soil NH₄-N and NO₃-N were determined at start and at 30 days by extraction with 2M KCl at a ratio of 1:5 (soil:KCl), and concentrations of each determined colorimetrically using a μ Quant™ micro-plate spectrophotometer (BioTek instruments, Inc. Winooski, VT). Contents of NaOH vials were titrated in the presence of BaCl₂ to a phenolphthalein end point with 1N HCl to determine excess NaOH. Carbon mineralization was determined using the formula shown below.

$$C = (B-V)NE \qquad \text{Eq [1]}$$

Where:

C = carbon in mg

B = acid needed to titrate NaOH in Blanks (mL).

V = acid needed to titrate the NaOH in sample vials (mL)

N = normality of acid and

E = carbon equivalent weight

Potential N mineralization was obtained as the difference in inorganic soil N between 0 and 30 days (Franzluebbers, 1999). Potential C mineralization was the CO₂-C

obtained after 30 days, corrected for blanks. Percent N mineralization and C turnover from amended soil were determined by dividing total C or N mineralized by total amended soil C or N, respectively. Total C or N was the summation of total soil and crop residue C and N, respectively.

Cover Crop Decomposition

Cover crop decomposition and mineralization was determined using nylon litter bags (10 x 20 cm) of mesh size 50-60 μm . Each bag received 5 Mg ha^{-1} grams oven dry weight equivalent of air-dried plant material before being sealed with an electric sealer and labeled with two tags; an aluminum tag and polythene paper covered tag. Details on the labels included cover crop identification, planned sampling time, placement method and location (landscape position). Polythene paper covered tags were color coded to represent sampling date for ease of identification during bag sampling. Bags were put in the field on 12 May 2005. The field was planted to corn in mid April 2005.

To simulate conservation (CsT) and conventional (CT) tillage, half of the bags were surface placed while the other half were buried for each plant material on each landscape position (landscape position development described above). For buried bags, soil was removed to 10 cm depth and bags were laid flat on the ground with 30-cm spacing between bags. Soil was replaced ensuring that the colored-coded paper tags were visible from the surface, while the aluminum tags were buried with the bags. The four plant materials were replicated four times across three landscape positions and sampled nine times during the six month period, (a total of 864 bags). Bag placement was

randomized within the plot, and sampling was done at 1, 2, 4, 6, 10, 14, 18, 22 and 26 weeks. At each sampling, a total of 96 bags were collected.

Sampled bags were cleaned of soil and other debris on arrival to the laboratory, and oven dried at 55°C to constant weight. Plant material remaining in each bag was weighed to determine mass loss. A subsample of the remaining material was ashed in muffle furnace at 450°C for 12 hours, and the mass remaining was expressed on an ash-free dry weight (AFDW) basis in order to correct for soil and other contamination. Subsamples of initial material were also ashed in the same way and their masses expressed on AFDW basis. The correction was made with a modified calculation from (Cochran, (1991):

$$\text{AFDW (g)} = \text{Dry weight (g)} - \text{Ash weight (g)} \quad \text{Eq [2]}$$

Ash free dry weight was used to obtain the percentage mass remaining at each sampling time using the formula:

$$\% \text{ Mass remaining} = \frac{\text{Final AFDW}}{\text{Initial AFDW}} \times 100 \quad \text{Eq [3]}$$

Phosphorus (P) and K content of the ashed material were determined by wet digestion (Hue and Evans, 1986) while total C and N were determined by dry combustion using LECO TruSpec CN analyzer.

Data Analysis

The generalized linear model (GLM) in SAS (SAS Inst., Cary, NC) was used to compare terrain attributes across landscape positions. Treatment means were compared using Fisher's protected least significant difference (LSD) at $P \leq 0.05$.

An asymptotic decay model was fitted to decomposition data using PROC NLMIXED in SAS to obtain mass, C and nutrient mineralization rate constants (k) and asymptotes (fraction of recalcitrant material or nutrient) for each plant material. This was accomplished for each placement method (buried versus surface placed) as well as landscape position by plotting mass or nutrient remaining versus time (days). The model was of the form shown below (Schaffers et al., 1998):

$$Y_{(t)} = a + (100-a)e^{-kt} \quad \text{Eq [4]}$$

Where $Y_{(t)}$ = the remaining fraction mass or nutrient at specific time t (days), a = the asymptotic remaining fraction mass or nutrient, and k (day^{-1}) = the rate at which the asymptotic fraction is approached. The model indicates that a fraction of the residue mass or nutrient will not decompose or will not be released during the time of study (Njunie et al., 2004). PROC GLM in SAS was used to compare decomposition and nutrient k of the cover crops. Treatment means were compared using Fisher's protected LSD at $P \leq 0.05$.

Due to alternating mineralization and immobilization of P from the crop residues, the asymptotic decay model could not be fit to the P data. Instead, mixed generalized linear model using PROC MIXED in SAS was used to account for sampling dates and to test for main effects and interactions. Treatment means were compared using least significant difference (LSD).

Stepwise regression in SAS was used to relate plant material chemical composition contribution to variation in mass loss and nutrient mineralization. In addition, correlation (in SAS) was used to relate P mineralization with initial chemical characteristics of the cover crop residues.

Cover crop residue mass loss k , C and N mineralization k , and K release k data were normalized (0-100) followed by principal component analysis in SAS. A plot of scores of the first two principal components was done to determine if there were distinct groupings relating to landscape and soil management treatments.

RESULTS

Landscape Variability

There were significant differences in landscape variability factors between landscape positions, except planimetric curvature and surface horizon silt content (Table 12). Sideslope landscape position had higher slope, surface horizon clay content and profile curvature (more convex shape) compared to the drainageway position. Higher slope on the sideslope result in runoff that would accumulate in the drainageway. This is depicted by higher flow accumulation, surface horizon sand content and compound topographic index (CTI) within the drainageway landscape. Positive profile curvature values found on the summit landscape position indicate a convex profile, while negative profile curvatures on drainageway indicate concave profiles (Li et al., 2005). Highest elevations and depth to seasonal high water table (SHWT) were found on the summit.

Potential C and N Mineralization

Chemical characteristics of cover crop residues used in this study are shown on Table 13. Crop residues had similar chemical properties, except black oat which had lower N content and higher neutral detergent fiber (NDF), lignin and hemicellulose.

There were significant treatment differences in potential N mineralization among soils amended with different crop residues ($P = 0.001$). Crimson clover and white lupin amended soil mineralized similar amounts of N (Table 14) that were higher than that mineralized from control soil. Spring forage rape amended soil mineralized similar amounts as the control soil. Black oat amended soil mineralized less N than the control soil at the start of the study, as well as at the end of 30 days, indicating net N immobilization (Table 14). Relative N mineralization in black oat amended soil was -0.48% of initial total N in the amended soil (soil + crop residue N), compared to 4% from crimson clover amended soil and 2% in control soil (Table 14). Net (less control soil N mineralization) N mineralization was in the order crimson clover = white = spring forage rape > black oat (Table 14). Stepwise regression indicated that crop residue NDF had negative effect on potential N mineralization and explained 67% of variability in potential N mineralization (data not shown).

Residue amended soil resulted in potential $\text{CO}_2\text{-C}$ mineralization patterns similar to N (Table 14). The C mineralization was, however, higher than that from control soil. Carbon turnover was similar in the four treatments, but higher than that of the control soil (Table 14). Black oat and crimson clover amended soil each mineralized 6.1% of initial total soil C (organic soil C + crop residue C), while spring forage rape and white lupin treated soil mineralized 6.3% compared to 1.3% from control soil. Stepwise regression using cover crop N, C, lignin, cellulose, hemicelluloses, NDF and C:N mineralized showed that none of the variables could explain significant variability in potential $\text{CO}_2\text{-C}$ mineralization (Data not shown).

Cover Crop Mass Loss and Nutrient Mineralization

Similar mass loss patterns were observed on the three landscape positions in both buried and surface placed materials (Fig. 18). Buried material mass loss rate was higher than that of surface placed materials during the initial decomposition phase. Fifty percent of the mass was lost within the first 10-20 days on buried materials, while the same loss was achieved after 50-75 days on surface placed materials. Once the easily decomposable material was mineralized, both buried and surface placed biomass decomposed at similar low rates (Fig. 18). A similar trend was observed for N mineralization and potassium (K) release rates (Fig. 19). Potassium release from incorporated residues was rapid, with at least 50% of K release occurring within the first 7 days, and 90% K release occurring by 10 days. Surface placed residues immobilized various amounts of N for at least the first 14 days (except black oat) (Fig. 19). Buried materials showed net N mineralization throughout the study period, except white lupin that immobilized N during the first 7 days. Buried white lupin immobilized about 4% of its initial N during the first 7 days, while surface applied white lupin immobilized N for the initial 28 days before net N mineralization occurred.

Since mass loss, C and nutrient mineralization rate constants (k) were similar on the three landscape positions, landscape positions were used as field replicates to compare k values using PROC GLM procedure in SAS. Results indicate that buried material had a significantly higher mass loss k than surface placed material ($P < 0.001$). Mean mass loss k of surface placed material was -0.017 day^{-1} , compared to -0.07 day^{-1} for buried materials (Table 15). Buried crimson clover, spring forage rape and white lupin had similar but higher mass loss k than black oat. Surface applied black oat had similar

mass loss k with surface placed white lupin and spring forage rape. This rate was significantly lower than that of crimson clover. Carbon mineralization k followed a similar trend as mass loss and was in the order crimson clover > spring forage rape = white lupin = black oat for surface placed materials (Table 16). Black oat had the lowest C k among buried materials. Stepwise regression of mass loss k against initial plant material chemical parameters showed effects of the chemical parameters on mass loss k differed depending on mode of material application. In buried materials, 82% of model variation was explained by NDF; an increase in NDF resulted in decreased mass loss k . In surface placed materials, 76% of regression model was explained by initial material cellulose content, with increasing cellulose resulting in decreased mass loss k .

Although N (and other nutrients) remaining in the litter bag at each sampling time were calculated by multiplying %N by mass remaining at the time of bag collection, patterns of N mineralization were different from those of mass loss. Nitrogen k was determined only for the period when net N mineralization occurred, since the asymptotic decay model could not adequately describe N immobilization. Buried cover crop residues had higher N mineralization k (-0.046 day^{-1}) than surface placed residues (-0.015 day^{-1}) (Table 17). Nitrogen mineralization k for buried residues was in the order crimson clover > spring forage rape > black oat > white lupin, while that of surface placed residues was crimson clover > = black oat > spring forage rape = white lupin (Table 17). Potassium release by the four residues was rapid achieving asymptotes early (Fig. 19) with k values ranging between -0.032 to -0.122 day^{-1} and -0.088 to 0.127 day^{-1} on surface placed and buried materials, respectively (Table 18). Buried materials had a significantly higher K mineralization rate constant than surface placed materials. In both material

placement methods, black oat and spring forage rape had higher K *k* compared to crimson clover and white lupin.

Phosphorus showed an initial rapid mineralization followed by alternating mineralization and immobilization after one month of sampling (Fig. 20). Due to this, the asymptotic model could not adequately describe P dynamics. Phosphorus mineralization was higher on the summit and in the drainageway landscape positions than on the sideslope position for surface applied residues (Appendix 6; Fig. 20). Buried residue mineralized P at similar rates on the three landscape positions. Also, spring forage rape residues mineralized P faster than other residues in both buried and surface placed materials (Fig. 21). Phosphorus mineralization of buried and surface placed crop residues positively correlated with initial P, K, C/P ratio and hemicellulose of the cover crop residues (Table 19). Negative correlation was observed between P mineralization and initial C concentration of the residues. Stepwise regression shows that initial residue K concentration explained 2% of the variance in P mineralization on buried materials, while initial residue P concentration contributed to 3% of the variability on surface applied residues. Regression relating P release to other variables is:

$$\% \text{ P released}_{(\text{buried})} = 55 + 4.9 \text{ K} + 0.2 \text{ NDF} + \varepsilon \quad R^2 = 0.028 \quad \text{Eq [5]}$$

$$\% \text{ P released}_{(\text{surface})} = -59 + 203 \text{ P} + 1.4 \text{ C} + 0.3 \text{ H} + \varepsilon \quad R^2 = 0.057 \quad \text{Eq [6]}$$

Where:

K = potassium

NDF = neutral detergent fiber

P = phosphorus

C = carbon, and

H = hemicellulose

ε = error term

Cover Crop Residue Decomposition and Mineralization Multivariate Analysis

Principal component analysis assigns eigenvalues to principal components which are ranked by their relative contribution to measured data variability. The first two principal components explained 91% of mass loss, C and N mineralization, and K release *k* variance. A plot of scores of the first two principal components shows mineralization data can be categorized into two groups based on crop residue application method (Fig. 22). No distinction can be made on landscape positions. Terrain attributes did not have significant effect on cover crop decomposition and mineralization *k* variability (from stepwise regression) for buried and surface applied residues.

DISCUSSION

Potential C and N Mineralization

Cover crops enhanced soil respiration as indicated by higher CO₂-C production from cover crop amended soil relative to control soil, suggesting cover crop residues provided substrate for microbial respiration. Relative C mineralization was similar for crimson clover, white lupin, spring forage rape and black oat amended soil, suggesting similarity in organic matter quality in the four amended soils. However, N mineralization patterns were different from those of C mineralization. Black oat amended soil immobilized N, suggesting that break down of its C occurred at the expense of N. It

would appear that black oat N concentration was not sufficient for microbial requirements in mineralization of C, and the microbes took up N from the soil resulting in net negative N mineralization. Higher potential N mineralization from crimson clover and white lupin amended soil compared to the control soil suggests improvement in soil organic N on addition of these residues.

Nitrogen immobilization in black oat amended soil contrasts with N dynamics observed on the same soil under field conditions in the litter bag study. In the field study, black oat residues mineralized N throughout the study period. Nitrogen immobilization in the laboratory study may be due to increased residue surface area available for microbial colonization, coupled with lower residue N concentration in black oat. The laboratory study utilized ground cover crop residues, while whole crop residue biomass was used in the field litter bag study. Bending and Turner (1999) found that effect of crop residue size on microbial respiration and N dynamics was dependent on the biochemical quality of the substrate and the stage of decomposition. Reducing residue particle size resulted in a delayed microbial respiration peak, and increased N immobilization in residues with high C:N ratios. They hypothesized that in high quality materials (hence fast decomposition), colonization of the residues by micro-biota is so rapid that surface area available for colonization has little effect on microbial activities. In our litter bag decomposition study (field), we established that black oat decomposed significantly slower (lower k) than crimson clover, spring forage rape and white lupin. Assuming that black oat would also decompose at a slower rate under laboratory conditions, increased surface area of ground material would result in increased microbial colonization resulting in net N immobilization. Similarly, Mary et al. (1996) observed

higher net N immobilization on addition of wheat straw under laboratory conditions than under field conditions. In our study, higher NDF content of black oat residues slowed decomposition of the residues and enhanced N immobilization in black oat amended soils.

Cover Crop Residue Mineralization

Cover crop residues used in this study were relatively similar in initial chemical parameters, resulting in similar mass loss rates (Table 13). Our results indicate faster decomposition occurred on incorporated material compared to surface applied material, and are similar to findings of Thonissen et al. (2000) and de Varennes et al. (2007). On the contrary, Abiven and Raceous (2007) found no significant effect of residue placement on C mineralization kinetics of mature crop residues under non-limiting soil N conditions. While other surface placed crop residues immobilized N during the first 14 days, black oat did not immobilize N when buried or surface placed. Among the four crop residues, black oat had the highest C:N ratio (23 compared to 11 in clover). Although Paul and Clark (1989) suggested that net N mineralization occurs when residue C:N ratio is less than 25, Trinsoutrot et al. (2000) observed initial N immobilization in materials with C:N ratios ranging from 10-150. In our study, immediate net N mineralization in buried materials was observed on black oat, crimson clover and spring forage rape crop residues, suggesting non-limiting N mineralization conditions. This may be explained in two ways: 1) favorable N mineralization conditions (namely moisture and temperature under buried conditions) (Thonissen et al., 2000) and 2) higher residue-soil contact that can result in higher microbial activity (de Varennes et al.,

2007). It is not clear why buried white lupin temporary immobilized N during the first seven days of the study.

Relatively higher NDF in black oat contributed significantly to its slower mass loss k compared to other residues. Similarly, Ruffo and Bollero (2003b) found that high NDF and acid detergent fiber (ADF) in winter cover crop residues resulted in low k . The two are components of cell walls and determine cell wall thickness that influences decomposition rate. White et al. (2004) found lower decomposition rates in plant materials high in NDF in a New Zealand grassland. Total initial residue N did not significantly influence decomposition k . Similar results were found by Trinsoutrot et al. (2000), who found only a weak correlation between decomposition of residues and residue organic N concentration at 7 days of incubation. They attributed this weak relationship to the high correlation between soluble C and organic N. Sarrantonio (2003) found that neither total residue N nor C:N ratio were reliable predictors of the quantity or percent of total N accumulated as NO_3 in soil amended with crop residues. According to Ruffo and Bollero (2003b), residue soluble N is more critical in controlling residue decomposition and C and N mineralization than total N.

At 7 days, about 50% loss in crop residue total P had occurred. This loss in residue P was much higher than mass loss (Fig. 18) and N mineralization (Fig. 19), but close to K loss on the same sampling date and may not be controlled by utilization of C by microbes (Salas et al. 2003). Instead, it may be related to crop residue inorganic P fraction that is easily released. This is further supported by low correlation between initial cover crop residue chemical composition and P release. Alternating mineralization and immobilization of P (Figs. 20 and 21) that occurred after 28 days may be explained

by the hypothesis put forward by McGill and Cole (1981), that P mineralization takes place through the action of extracellular enzymes. Production of these enzymes is controlled by amount of P in soil solution in response to microbial need for P, as opposed to organic P mineralization during microbial SOM C oxidation for energy. An increase in P concentration in the residues would be related to microbial immobilization as well as a decrease in extracellular enzyme activity (Joann et al., 2001).

Stepwise regression showed no significant effect of terrain attributes on cover crop decomposition and mineralization k for buried and surface applied residues (data not shown).

Cover Crop Residue Mineralization Principal Component Analysis

Lack of differences in cover crop residue mass loss and nutrient mineralization on the three landscape positions suggest that landscape variability may not have been sufficient to cause differences in cover crop decomposition and mineralization. This is further supported by principal component analysis results. The first two principal components explained most (91%) of the data variability in mass loss and mineralization k . Scores of the first two principal components categorize cover crop decomposition and mineralization k based on residue placement method irrespective of landscape position (Fig. 22). Surface residue application treatment represents CsT, while buried residues represent CT treatment. Thus overall, cover crop residue decomposition was mainly influenced by tillage system.

CONCLUSIONS

Buried cover crop residues lost mass and mineralized faster than surface applied residues, suggesting that crop residues under CT would decompose and mineralize faster than those under CsT systems. The initial rapid mass loss was followed by a slow loss that was similar for buried and surface placed cover crops. This was observed on the three landscape positions, with no cover crop decomposition or nutrient mineralization differences observed between landscape positions.

All buried cover crops decomposed at similar rates, except black oat, which decomposed slower than other cover crops. Surface placed spring forage rape and crimson clover decomposed faster than black oat and white lupin residues. Slower decomposition rate of black oat was due to higher NDF content of the material. Overall, cover crop residue decomposition and mineralization were influenced by residue application method.

Faster mass and nutrient loss from incorporated residues compared to surface placement has implications on nutrient availability to subsequent crops. If cover crop residues such as those used in this study are to be useful in crop nutrition, maintaining residues at the surface would be advisable to minimize nutrient losses that may occur upon rapid decomposition if incorporated. Such cover crops may need to be terminated a few weeks before crop planting to enhance synchrony between nutrient release and crop demand.

Slower mass loss and nutrient release from decomposing surface applied cover crop residues mulch the soil surface and result in weed suppression through competition and shading. The mulch also conserves soil moisture by reducing evaporation from the

soil. Slower decomposing cover crops also contribute to soil organic matter buildup compared to fast decomposing legume residues. Among the cover crops tested, black oat would be the best choice for weed suppression, soil moisture conservation.

Landscape variability appears to have been insufficient to cause differences in cover crop decomposition and mineralization.

Table 12. Analysis of variance of terrain attributes among summit, drainageway and sideslope landscape positions at E.V. Smith Research Center near Shorter, AL.

Terrain attribute	Soil landscape position†			P-value
	Summit	Sideslope	Drainageway	
CTI‡	3.98b	4.15b	6.16a	0.0001
Elevation (m)	71.33a	70.53b	69.49c	0.0001
Planimetric curvature	0.01a	-0.01ab	-0.08b	0.0620
Profile curvature	0.02a	0.02a	-0.09b	0.0030
Slope (%)	0.60c	3.33a	1.33b	0.0001
Flow accumulation	05.01b	7.13b	30.35a	0.0040
SHWT (cm)§	145.83a	108.33b	75.00c	0.0010
Sand (%)	56.78b	54.32b	63.75a	0.0001
Silt (%)	24.44a	25.50a	25.20a	0.7360
Clay (%)	18.79a	21.12a	11.06b	0.0001

† Values in rows followed by the same letter are not significantly different at $P \leq 0.05$.

‡ Compound topographic index

§ Seasonal high water table

¶ Surface horizon sand, silt and clay content

Table 13. Selected characteristics of four cover crop residues used in the decomposition study

Residue	C	N	C:N	NDF†	Lignin	Cellulose	Hemicellulose		
	----- g kg ⁻¹ -----								
Black oat			392	17	23	470	46	230	193
Crimson clover			410	38	11	273	30	169	74
White lupin			417	21	20	262	31	216	16
Spring forage rape			368	27	14	229	25	197	07

† Neutral detergent fiber

Table 14. Potential C and N mineralization of cover crop amended soil. Soil was from conventionally tilled treatments at the 9-ha experiment at E.V. Smith Research Center near Shorter, AL.

Treatment	Potential C mineralization	Potential N mineralization	Net potential N mineralization ‡	Relative N mineralization	C turnover
	----- (mg kg ⁻¹ soil) -----			----- % -----	
Black oat	570	-4.7	-17.5	-0.5	6.1
Crimson clover	571	29.1	-16.3	3.9	6.1
Control soil	107	12.8	-	2.0	1.3
White lupin	591	24.7	11.8	3.4	6.3
Spring forage rape	582	17.6	04.7	2.5	6.3
ANOVA					
P-value	0.001	0.001	0.023	0.002	0.001
LSD†	75	11.7	12.4	1.6	0.8

† Least significant difference at P ≤ 0.05

‡ Control soil N mineralization was subtracted from N mineralization in cover crop amended soil

Table 15. Decay parameters from asymptote model describing mass loss of cover crops. Asymptote estimate indicates fraction of initial mass (%) that will not be decomposed within the time frame of study.

Residue placement†	Residue name	Asymptote estimate	Mass loss $k‡$ (day ⁻¹)
Surface	Black oat	20.21	-0.014
Surface	Crimson clover	20.94	-0.022
Surface	spring forage rape	16.96	-0.017
Surface	White lupin	21.34	-0.017
Buried	Black oat	17.10	-0.050
Buried	Crimson clover	16.42	-0.077
Buried	spring forage rape	15.65	-0.078
Buried	White lupin	20.08	-0.075

ANOVA

Effect	P-value	LSD§
Residue placement effect	0.0005	0.014
Crop residue effect (surface)	0.0004	0.003
Crop residue effect (buried)	0.0001	0.008

† Surface placed litter bags were secured with pins on the soil surface while buried litter bags were placed 10 cm below the soil surface.

‡ Rate constant

§ Least significant difference at $P \leq 0.05$

Table 16. Decay parameters from asymptote model describing C mineralization of cover crops. Asymptote estimate indicates fraction of initial C (%) that will not be released within the time frame of study.

Residue placement†	Residue name	Asymptote estimate	C mineralization $k‡$ (day ⁻¹)
Surface	Black oat	13.49	-0.012
Surface	Crimson clover	17.38	-0.021
Surface	Spring forage rape	13.21	-0.016
Surface	White lupin	17.36	-0.015
Buried	Black oat	16.15	-0.045
Buried	Crimson clover	15.38	-0.077
Buried	Spring forage rape	15.13	-0.080
Buried	White lupin	19.86	-0.070

ANOVA

Effect	P-value	LSD§
Residue placement effect	0.0001	0.0093
Crop residue effect (surface)	0.0001	0.0028
Crop residue effect (buried)	0.0004	0.0154

† Surface placed litter bags were secured with pins on the soil surface while buried litter bags were placed 10 cm below the soil surface.

‡ Rate constant

§ Least significant difference at $P \leq 0.05$

Table 17. Decay parameters from asymptote model describing N mineralization of cover crops. Asymptote estimate indicates fraction of initial N (%) that will not be released within the time frame of study.

Residue placement†	Residue name	Asymptote estimate	N mineralization $k‡$ (day ⁻¹)
Surface	Black oat	40.88	-0.013
surface	Crimson clover	20.68	-0.024
Surface	Spring forage rape	14.94	-0.010
Surface	White lupin	39.43	-0.009
Buried	Black oat	32.33	-0.032
Buried	Crimson clover	16.81	-0.078
Buried	Spring forage rape	21.17	-0.050
Buried	White lupin	35.44	-0.023

ANOVA		
Effect	P-value	LSD§
Residue placement effect	0.0001	0.011
Crop residue effect (surface)	0.0087	0.008
Crop residue effect (buried)	0.0001	0.014

† Surface placed litter bags were secured with pins on the soil surface while buried litter bags were placed 10 cm below the soil surface.

‡ Rate constant

§ Least significant difference at $P \leq 0.05$

Table 18. Decay parameters from asymptote model describing K mineralization of cover crops. Asymptote estimate indicates fraction of initial K (%) that will not be released within the time frame of study.

Residue placement†	Residue name	Asymptote estimate	K release $k‡$ (day ⁻¹)
Surface	Black oat	24.51	-0.122
Surface	Crimson clover	7.74	-0.032
Surface	Spring forage rape	18.97	-0.099
Surface	White lupin	14.68	-0.035
Buried	Black oat	3.38	-0.103
Buried	Crimson clover	1.95	-0.098
Buried	Spring forage rape	1.67	-0.127
Buried	White lupin	2.80	-0.088

ANOVA		
Effect	P-value	LSD§
Residue placement effect	0.002	0.018
Crop residue effect (surface)	0.005	0.047
Crop residue effect (buried)	0.056	0.028

† Surface placed litter bags were secured with pins on the soil surface while buried litter bags were placed 10 cm below the soil surface.

‡ Rate constant

§ Least significant difference at $P \leq 0.05$

Table 19. Correlation coefficients relating cover crop residue P mineralization with initial chemical composition of the cover crop residues.

Dependent variable	Independent variable	Correlation coefficient	P-value
% Phosphorus released	N	0.043	0.207
	P	0.117	0.001
	K	0.134	0.001
	C	-0.071	0.038
	NDF	0.077	0.247
	ADF	-0.011	0.753
	lignin	0.063	0.066
	Hemicellulose	0.101	0.003
	Cellulose	-0.035	0.302
	C:N	-0.034	0.317
	C:P	-0.117	0.001

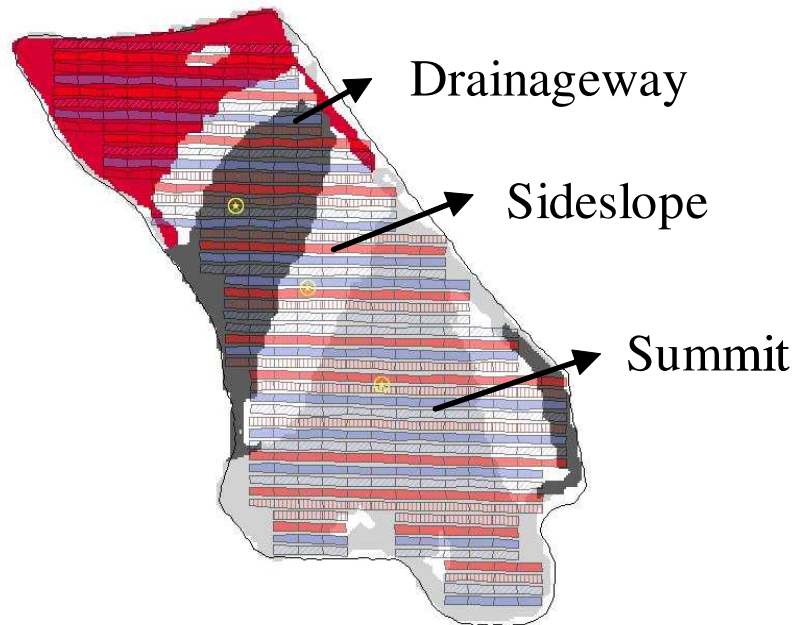


Fig. 17. Study site soil landscape positions created using fuzzy k-means unsupervised clustering based on seasonal high water table, digital elevation, slope and compound topographic index. Summit is the highest position, drainageway the lowest position, while sideslope is an eroded landscape.

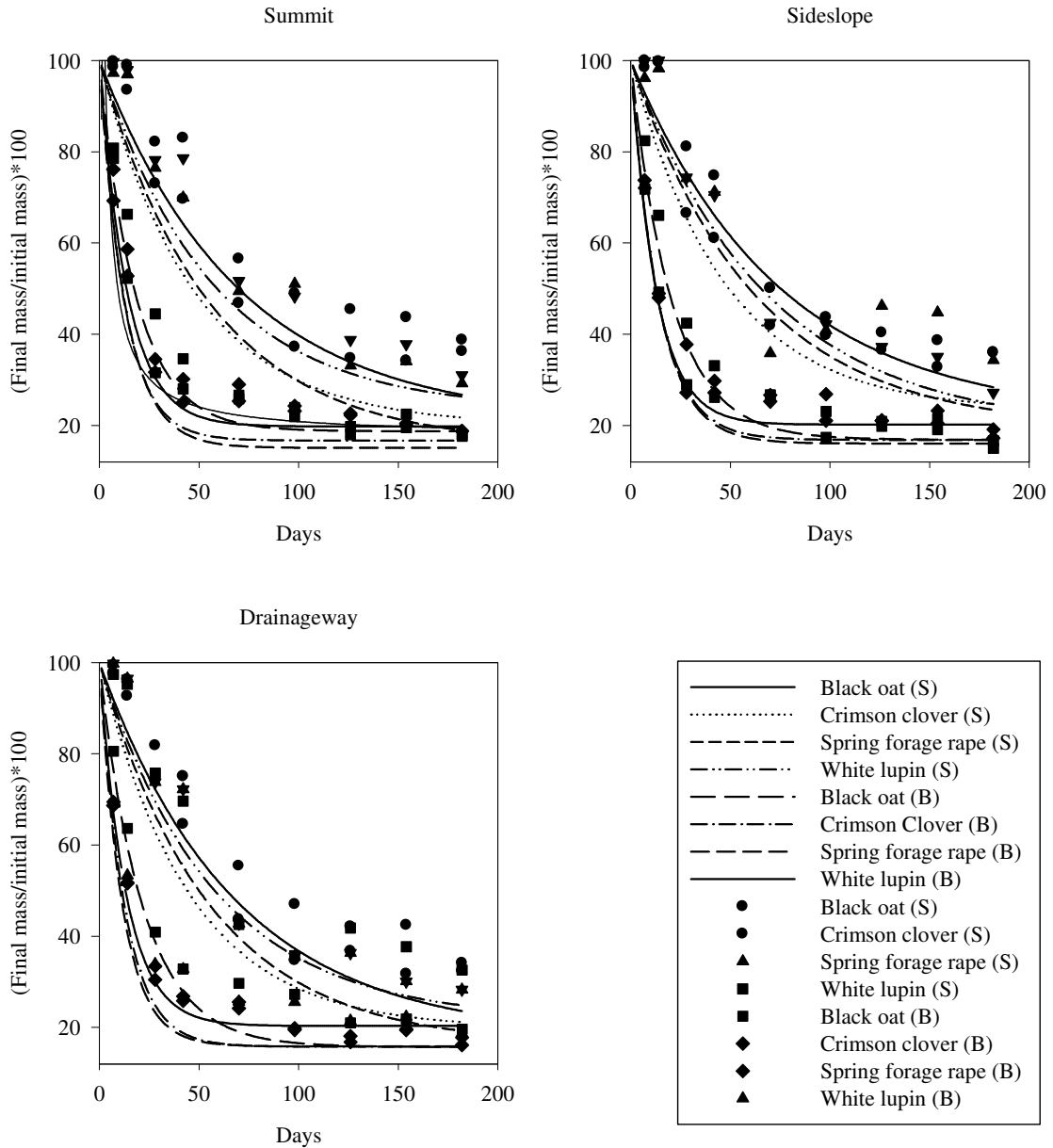


Fig. 18. Mass loss of surface and buried cover crop residues across three soil landscape positions. Scatter plots are measured data while lines are derived from asymptote decay prediction model. S= surface crop residue placement, B = buried crop residue.

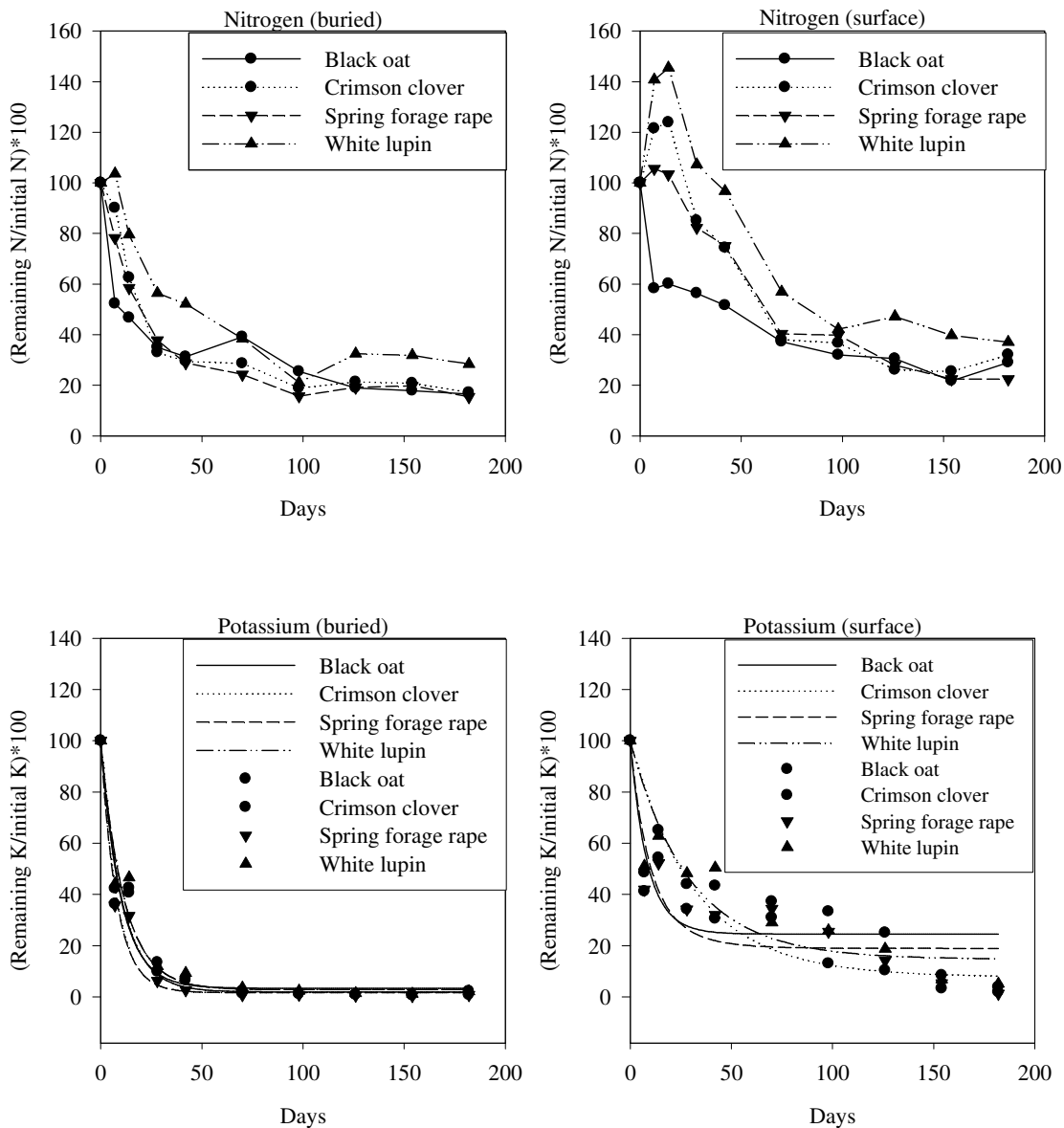


Fig. 19. Nitrogen mineralization and potassium (K) release of buried and surface placed cover crop residues (landscape position means). Nitrogen data was obtained from nutrient determination of residue remaining at each sampling time. Scatter plots represent K measured data while lines represent output from asymptote model.

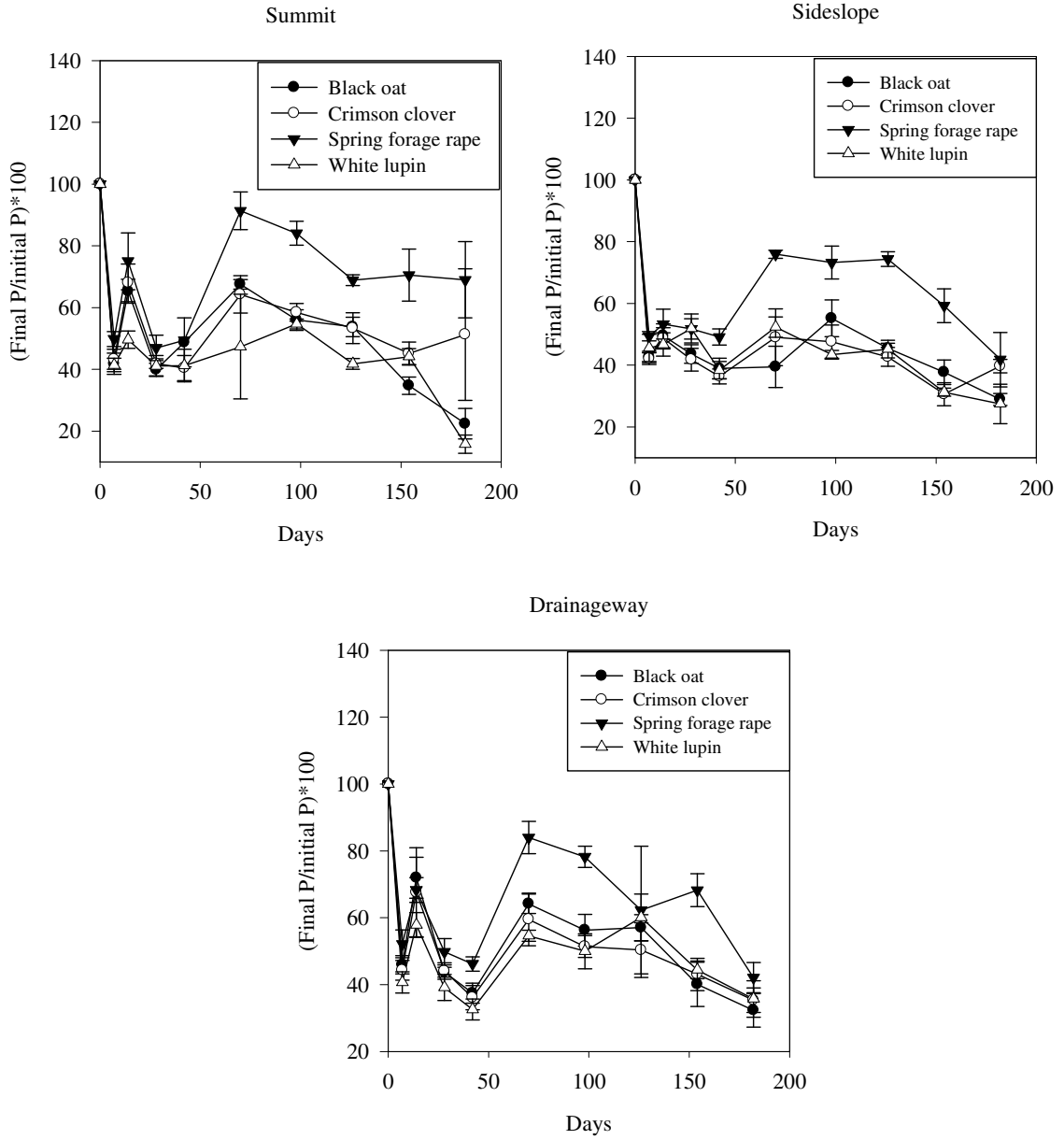


Fig. 20. Phosphorus mineralization of surface placed cover crop residues on each landscape position. Bars are standard errors of the mean.

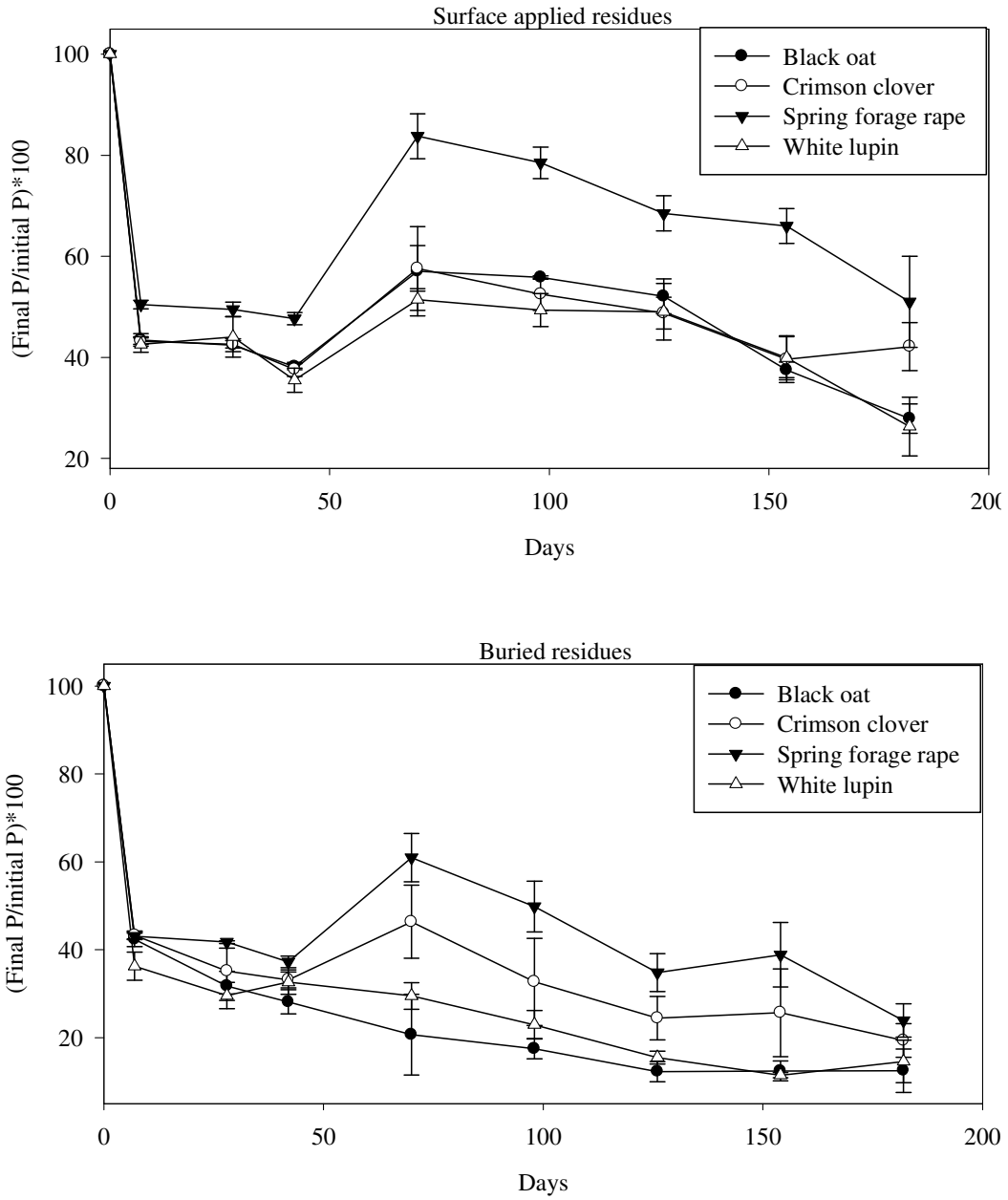


Fig. 21. Phosphorus mineralization of surface placed and buried cover crop residues averaged across landscape positions. Bars are standard errors of the mean.

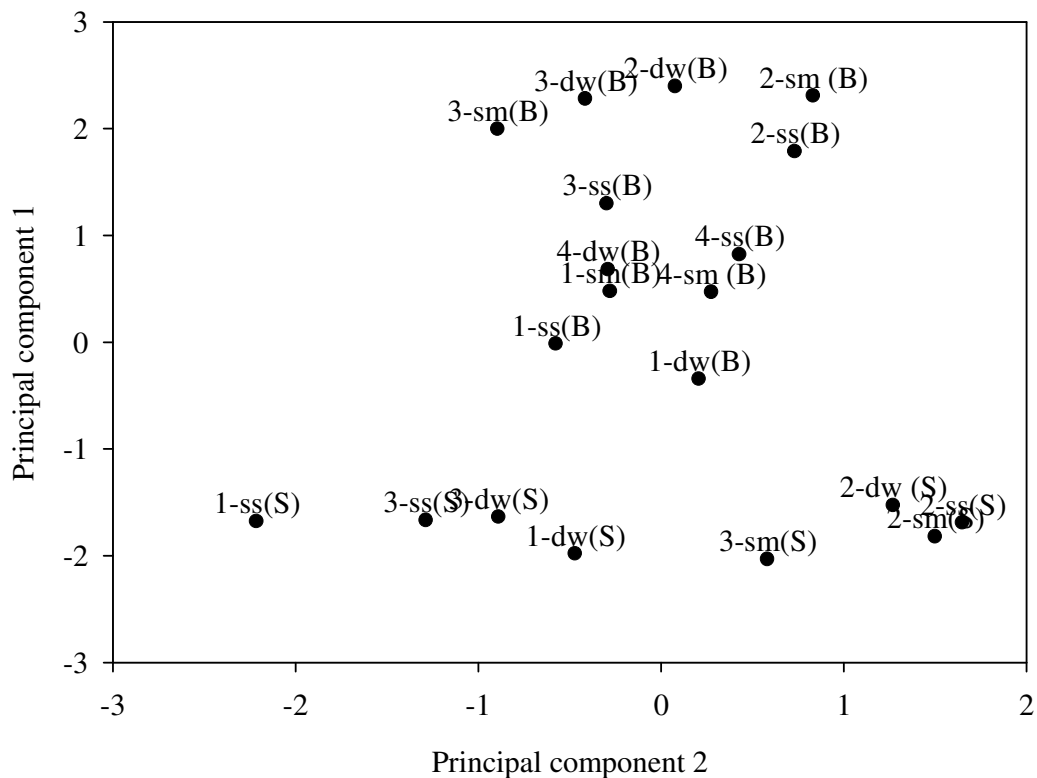


Fig. 22. Decomposition rate constants (k) principal component scores of the first two principal components. Data includes mass loss, C and N mineralization, and K release k . 1= black oat, 2 = crimson clover, 3 = spring forage rape and 4 = white lupin. sm = summit position, ss = sideslope position, dw = drainageway position, S= surface residue application, B = buried residue application.

V. SUMMARY

Carbon and N dynamics, greenhouse gas emissions, and cover crop decomposition are influenced by soil management, soil landscape variability, and environmental factors. Three landscape positions (summit, sideslope, drainageway) were delineated at the field-scale on a 9-ha experimental site at the E.V. Smith Research Center. The site is typical for Coastal Plain landscapes of the southeastern US. The summit position has higher elevation and a deeper SHWT, while the drainageway is higher in flow accumulation and compound topographic index. The sideslope landscape position has a higher slope, surface horizon clay content, and profile curvature. On each landscape position, the effects of tillage and dairy manure on CH₄, N₂O and CO₂ emissions as well as C and N dynamics were determined. Variability in gas emissions and C and N dynamics were best explained by management, but terrain attributes contributed substantially to these variations.

The summit landscape position oxidized CH₄, while the drainageway emitted CH₄. Methane emission was increased by dairy manure application across the three landscape positions. Soil management affected N₂O emission in the drainageway, but not on the summit and sideslope. In seasons when soil treatment differences occurred within the drainageway, dairy manure application decreased N₂O emissions. This decrease was perhaps due to increased soil moisture that may have resulted in reduction of N₂O to N₂.

Conventional tillage treatments tended to have lower soil moisture and higher N₂O emissions compared to CTM treatments. Carbon dioxide emission was higher on CsT treatments than in CT treatments during the winter seasons. No soil management or landscape differences were observed in CO₂ emissions in spring, summer and fall seasons.

Conservation tillage and use of dairy manure increased soil organic C and total N following the six years of treatment application. This effect was similar on the three landscape positions. Dairy manure increased soil organic C and total soil N in both CT and CsT systems. C:N mineralized, and relative C mineralization were similar in the four treatments suggesting similarity in soil organic matter quality between managements.

Potential C mineralization was similar for crimson clover, spring forage rape and white lupin amended soil. White lupin and crimson clover amended soil mineralized higher N than control soil, but spring forage rape amended soil mineralized similar amounts of N as the control soil. Black oat amended soil immobilized N under laboratory conditions, but mineralized N in the field (decomposition study). The difference in this observation may be explained by difference in residue size and residue application methods used in the field compared with laboratory. Decomposition and mineralization of the cover crop residues under field conditions was unaffected by soil landscape position. Crimson clover, white lupin and spring forage rape decomposed and mineralized faster than black oat. Potassium release was rapid, but P mineralization was erratic alternating between immobilization and mineralization.

Terrain attributes did not significantly influence soil C and N dynamics relative to CH₄, N₂O and CO₂ emissions. We emphasize that gas measurements were not taken immediately following tillage and dairy manure application. Rather, the measurements were taken several weeks and sometimes several months after field operations, depending on the season.

Overall, soil management had greater impact on C and N dynamics and cover crop decomposition than landscape variability. Although terrain attributes influenced these processes, soil management had greater impact on C and N mineralization, CH₄, N₂O and CO₂ fluxes, and cover crop decomposition than landscape variability.

LITERATURE CITED

- Abiven, S. and S. Raceous. 2007. Mineralization of crop residues on the soil surface or incorporated in the soil under controlled conditions. *Soil Biol. Fertil.* 43:849-852.
- Ai-Zhen, L. Zang, X., Fang, H., Yang X and C.F. Drury. 2007. Short-term effects of tillage practices on organic carbon in clay loam soil of northeast China. *Pedosphere* 17(5):619-623.
- Al-Kaisi, M.M. and X. Yin. 2005. Tillage and crop residue effects on soil carbon and carbon dioxide emission in corn-soybean rotations. *J. Environ. Qual.* 34:437-445.
- Al-Kaisi, M.M., M.L. Kruse and J.E. Sawyer. 2008. Effect of nitrogen fertilizer on growing soil carbon dioxide emission in a corn-soybean rotation. *J. Environ. Qual.* 37:325-332.
- Anderson, J.M., and J.S. Ingram. 1989. *Tropical Soil Biology and Fertility: A handbook of Methods.* CAB Wallingford, Oxford.
- Ball, B.C., A. Scott, and I.P McTaggart. 2002. Greenhouse gas emissions under organic management. p. 243-246. *In* Powell (ed), UK organic research. Proceedings of the COR conference, Aberystwyth.
- Bending, G.D., and Turner M. K. 1999. Interaction of biochemical quality and particle size of crop residues and its effect on the microbial biomass and nitrogen dynamics following incorporation into the soil. *Biol. Fertil. Soils* 29(3):319-327.
- Boeckx, P., O. Van Clement, and I. Villaralvo. 1996. Methane emission from a landfill and the methane oxidizing capacity of its covering soil. *Soil Bio. Biochem.* 28(10):1397-1405.
- Boeckx, P., and O. Van Clement. 1996. Methane oxidation in a neutral landfill cover soil: influence of moisture content, temperature, and nitrogen turnover, *J. Environ. Qual.* 25:178-183.

- Brady, N.C. and R.R. Weil. 2002. *The Nature and Properties of soils*. Prentice Hall, Upper Saddle River, NJ.
- Breuer, L., R. Kiese, and K. Butterbach-Bahl. 2002. Temperature and moisture effects on nitrification rates in tropical rain-forest soils. *Soil Sci. Soc. Am. J.* 66:834-844.
- Calderon, F.J. and Jackson L.E. 2002. Rototillage, disking, and subsequent irrigation: Effects on soil nitrogen dynamics, microbial biomass, and carbon dioxide efflux. *J. Env. Qual.* 31:352-358.
- Chan, A.S.K. and T.B. Parkin. 2001. Effect of land use on methane flux in soils. *J. Environ. Qual.* 30:786-797.
- Cicerone, R.J. 1987. Changes in stratospheric ozone. *Science* 237:35-42.
- Clayton, H., I.P. McTaggart, J. Parker, L. Swan, and K.A. Smith. 1997. Nitrous oxide emissions from fertilised grassland: a 2 year study of the effects of N fertiliser form and environmental conditions. *Biol. Fertil. Soils* 25: 252-260.
- Cochran, V.L. 1991. Decomposition of barley straw in a sub-arctic in the field. *Biol. Fertil. Soils* 10:227-232.
- Cole, C.V., J. Duxbury, J. Freney, O. Heinermeyer, K. Minami, A. Mosier, K. Paustian, N. Rosenberg, N. Sampson, D. Sauerbeck, and Q. Zhao. 1997. Global estimates of potential mitigation of greenhouse gas emissions by agriculture. *Nutr. Cycl. Agroecosyst.* 49:221-228.
- Collins, A. and D.W. Allison. 2002. Nitrogen mineralization in soil from perennial grassland measured through long-term laboratory incubations. *J. Agric. Sci.* 138:301-310.
- Cueto-Wong, J.A., S.J. Guldan, W.C. Lindeman, and M.D Remmenga. 2001. Nitrogen recovery from ¹⁵N-labeled green manures: recovery by forage sorghum and soil one season after green manure incorporation. *J. Sustain. Agric.* 17(4):27-42.
- Davidson, E.A. 1992. Sources of nitric and nitrous oxide following wetting and drying of dry soil. *Soil. Sci. Soc. Am. J.* 56:95-102.

- De Varennes, A., M.O Torres, C. Cunha-Queda, M.J. Goss and C. Carranca. 2007. Nitrogen conservation in soil and crop residues as affected by crop rotation and soil disturbance under Mediterranean conditions. *Biol. Fertil. Soils* 44:49-58.
- Dhima, K.V., I.B. Vasilakoglou, I.G. Eleftherohorinos, and A.A. Lithourgidis. 2006. Allelopathic potential of winter cereal cover crop mulches on grass weed suppression and sugar beet development. *Crop Sci.* 46:1682-1691.
- Dickinson, R.E. and R.J. Cicerone. 1986. "Future Global Warming from Atmospheric Trace Gases." *Nature (London)* 319:109-15.
- Dinesh, R., M.A Suryabarayanam, S.G. Chaudhuri and T.E. Sheeja. 2003. Long-term influence of leguminous cover crops on the biochemical properties of a sandy clay loam Fluventic Sulfaquent in a humid tropical region of India. *Soil Tillage Res.* 77:69-77.
- Farrel, R., D. Pennock, J.D., Knight and B. S. 2003. Quantifying N₂O fluxes associated with agricultural practices in non-level prairie landscapes. CCFIA Component 2-Project No. 3. Canadian Agri-Food Research Council. Climate Change Funding Initiative in Agriculture.
- Fisk, J. W., O.B. Hesterman, A. Shrestha, J. J. Kells, R. R. Harwood, J.M. Squire, and C.C. Sheafter. 2001. Weed suppression by annual legume cover crop in no-tillage corn. *Agron. J.* 93:319-329.
- Frank, D.A. and P.M. Groffman. 1998. Ungulate vs. landscape control of soil and N processes in grasslands on Yellowstone National Park. *Ecology* 79(7):2229-2241.
- Franzluebbers, A.J. 1999. Potential C and N mineralization and microbial biomass from intact and increasingly disturbed soils of varying texture. *Soil Biol. and Biochem.* 31: 1083-1090.
- Franzluebbers, A.J. 2005. Soil organic carbon sequestration and agricultural greenhouse gas emissions in the southeastern USA. *Soil Tillage Res.* 83:120-147.
- Freney, J.R., O.T. Denmead, and J.R. Simpson. 1978. Soil as a source or sink for atmospheric nitrous oxide. *Nature (London)* 273:530-532.
- Fridgen, J.J., N.R. Kitchen, K.A. Sudduth, S.T. Drummond, W.J. Wiebold, and C.W. Fraisse. 2004. Landscape position Analyst (MZA): Software for subfield landscape position delineation. *Agron. J.* 96:100-108.

- Gregorich, E.G., P. Rochette, S. McGuire, B.C. Liang and R. Lessard. 1998. Soluble organic carbon and carbon dioxide fluxes in maize fields receiving spring-applied manure. *J. Environ. Qual.* 27:209-214.
- Gilliam, F.S., N.L. Lyttle, A. Thomas, and M. B. Adams. 2005. Soil variability along a nitrogen mineralization-saturated hardwood forest. *Soil Sci. Soc. Am. J.* 69:247-254.
- Ginting, D., A. Kassevalou, B. Eghball and J.W. Doran. 2003. Greenhouse gas emissions and soil indicators four years after manure and compost applications. *J. Environ. Qual.* 32:23-32.
- Hall, S.J and P.A. Matson. 2003. Nutrient status of tropical rain forests influences soil N dynamics after N additions. *Ecol. Monogr.* 73(1): 107-129.
- Haney, R.L., F.M. Hons, M.A. Anderson and A.J. Franzluebbbers. 2001. A rapid procedure for estimating nitrogen mineralization in manured soil. *Boil. Fertil. Soils* 33:100-014.
- Haney, R.L., A. . Franzluebbbers, E.B. Porter, F.M. Hons and D.A. Zuberer. 2004. Soil carbon and nitrogen mineralization: influence of drying temperature. *Soil Sci. Soc. Am. J.* 68:489-492.
- Hashimoto, S., and H. Komatsu. 2005. Relationship between CO₂ concentration and CO₂ production, temperature, water content, and gas diffusivity: implications for field studies through sensitivity analysis. *J. For. Res.* 11:41-50.
- Huang, Y., J. Zou , X. Zheng, Y. Wang, and X. Xu. 2004. Nitrous oxide emissions as influenced by amendment of plant residues with different C:N ratios. *Soil Biol. Biochem.* 36:973-981.
- Hue, N.V., and C.E. Evans. 1986. Procedures used for soil and plant analysis by the Auburn University Soil Testing Laboratory. Dept. of Agron. & Soils, Auburn Univ. Dept. Series 106.
- Hussain, I., K.R. Olson and S.A. Ebelhar. 1999. Long-term tillage effects on soil chemical properties and organic matter fractions. *Soil Sci. Soc. Am. J.* 63:1335-1341.
- Hütsch, B.W. 1997. Tillage and land use effects on methane oxidation rates and their vertical profiles in soil. *Biol. Fertil. Soils* 27:284-292.

- Hütsch, B.W. 1998. Methane oxidation in arable soil as inhibited by ammonium, nitrate and organic manure with respect to soil pH. *Biol. Fertil. Soils* 28:27-35.
- Hütsch, B.W. 2001. Methane oxidation in non-flooded soils as affected by crop production- invited paper. *European J. Agron.* 14(4):237-260.
- IPCC (Intergovernmental Panel on Climate Change), 2001. *Climate change 2001. The Scientific Basis.* Cambridge University Press, Cambridge.
- IPCC (Intergovernmental Panel on Climate Change). 1990. *Climate change: The IPCC scientific assessment.* J.T. Houghton (ed.) Cambridge Univ. Press, Cambridge.
- Jacinthe, P.A., R. Lal and J.M. Kimble. 2002. Carbon budget and seasonal carbon dioxide emission from the central Ohio Luvisol as influenced by wheat residue amendment. *Soil Tillage Res.* 67(2):147-157.
- Jacob, D.J. 1999. *Introduction to atmospheric chemistry.* Princeton University Press, NJ.
- Joann, K.W., C. Chang and B.M. Olson. 2001. Nitrogen and phosphorus mineralization potentials of soils receiving repeated annual cattle manure applications. *Biol. Fertil. soils* 3(5):334-341.
- Karen, E.D, and K.A. Smith. 2003. Nitrous oxide emission factors for agricultural soils in Great Britain: the impact of soil water-filled pore space and other controlling variables. *Glob. Change Biol.* 9(2): 204-218.
- Keller, M., E.M. Mitre and R.F. Stallard. 1990. Consumption of atmospheric methane in soils of central Panama: effects of agricultural development. *Global Biogeochem. Cycle* 4:21-27.
- Kenneth, L.M., J.M. Klopatek and C.C. Klopatek. 1998. The effects of litter quality and climate on decomposition along an elevation gradient. *Ecol. Appl.* 8(4):1061-1071.
- Kim, D., and J. Kim. 2002. Soil nitric and nitrous oxide emissions from agricultural and tidal flat fields in southwestern Korea. *J. Environ. Eng. Sci.* 1:359-369.
- Kingery, W.L., C.W. Wood and J.C. Williams. 1996. Tillage and amendment effects on soil carbon and nitrogen mineralization and phosphorus release. *Soil & Tillage Res.* 37:239-250.

- Lee, J., J. Six, A.P. King, C. van Kessel, and D.E. Rolston. 2006. Tillage and field scale controls on greenhouse gas emissions. *J. Environ. Qual.* 35:714-725.
- Lee, D.K., J.J. Doolittle and V.N. Owens. 2007. Soil carbon dioxide fluxes in established switchgrass land managed for biomass production. *Soil Biol. Biochem.* 39:178-186.
- Le Mer, J., and P. Roger. 2001. Production, oxidation, emission and consumption of methane by soils: a review. *Eur. J. Soil Biol.* 37:25-50.
- Li, Z., Qing Z. and Christopher G. 2005. Digital terrain modeling principles and methodology. CRC Press, London.
- Liu, A., B.L. Ma, and A.A. Bomke. 2005. Effects of cover crops on soil aggregate stability, total organic carbon, and polysaccharides. *Soil Sci. Soc. Am. J.* 69:2041-2048.
- Lou, Y., Ren L., Z. Li and T. Zhang. 2007. Effect of rice residues on carbon dioxide and nitrous oxide emissions from a paddy soil of subtropical China. *Water Air Soil Pollut.* 178:157-168.
- Lowe, D. C. 2006. Global change: A green source of surprise. *Nature (London)* 439:148-149.
- Ma, L., G.A. Peterson, L.R. Ahuja, L. Sherrod, M.J. Shaffer, and K.W. Rojas. 1999. Decomposition of surface crop residues in long-term studies of dry land agroecosystems. *Agron. J.* 91:401-409.
- Mary, B., S. Raceous, D. Darwins and D. Robin. 1996. Interactions between decomposition and plant residues and nitrogen cycling in soil. *Plant and Soil* 182:71-82.
- McGill, W.B and C.V. Cole 1981. Comparative aspects of cycling of organic C, N, S and P through soil organic matter. *Geoderma* 26:267-286.
- McSwiney, C.P., W.H., McDowell and M. Keller. 2001. Distribution of nitrous oxide and regulators of its production across a tropical rainforest catena in the Luquillo Experimental Forest, Puerto Rico. *Biogeochemistry* 56:256-286.
- Meixner, X.F., and W. Eugster. 1999. Effects of landscape patterns and topography on emissions and transport. *In: Tecthunen, J.D. and P. Kabat (eds) 1999.*

Integrating hydrology, ecosystem dynamics, and biogeochemistry in complex landscapes series report of the Dahlem workshop held in Berlin, January 18-23, 1998. John Wiley & sons, Chichester.

Mitchell, J. 2002. Decomposition and nutrient release dynamics of cover crop materials. Project report. Western region SARE. SW97-045. Parlier, CA

Moore, I.D., P.E. Glessker, G.A. Nielsen, and G.A. Peterson. 1993. Soil attribute prediction using terrain analysis. *Soil Sci. Soc. Am. J.* 57:443-452.

Morris, S.J., and R.E.J. Boerner. 1998. Landscape patterns of nitrogen mineralization and nitrification in southern Ohio hardwood forest. *Landscape Ecol.* 13:215-224.

Mosier, A.R., and D.S. Schimel. 1991. Influence of agricultural nitrogen on atmospheric methane and nitrous oxide. *Chem. Ind.* 23: 874-877.

Mossier, A.R., C. Kroeze, C. Navison, O. Oenema, S. Seitzinger, O. Van Cleemput. 1998. Closing the global N₂O budget: Nitrous oxide emissions through the agricultural nitrogen cycle. *Nutr. Cycling in Agroecosyst.* 52:225-248.

Nadelhoffer, K.J. 1990. Microlysimeter for measuring nitrogen mineralization and microbial respiration in aerobic soil incubations. *Soil Sci. Soc. Am. J.* 54:411-415.

Neil, C., M.C. Piccolo, C.C. Ceri, P.A. Steudler, J.M. Mellilo, and M. Brito 1997. Net nitrogen mineralization and net nitrification rates in soils following deforestation for pasture across southwestern Brazilian Amazon basin landscape. *Ecologia* 110:243-252.

Njunie, M.N., M.G. Wagger and P. Luna-Orea. 2004. Residue decomposition and nutrient release dynamics from two tropical forage legumes in a Kenyan environment. *Agron. J.* 96:1073-1081.

Odhambo, J.J.O., and A.A. Bomke. 2000. Short term nitrogen availability following overwinter cereal/grass and legume cover crop monocultures and mixtures in south coastal British Columbia. *J. Soil and Water Conservation.* 55(3):347-354.

Oliver, D.J., Bloem, A. Vos and J.C Munch. 2004. Bacterial diversity on agricultural soils during litter decomposition. *Appl. Environ. Microbiol.* 70(1):468-474.

- Oorts, K., B. Nicholardot, R. Marckx, G. Richard and H. Boizard. 2006. C and N mineralization of undisrupted and disrupted soil from different structural zones of conventional tillage and no-till systems in northern France. *Soil Biol. Biochem.* 38:2576-2586.
- Pathak, H. 1999. Emissions of nitrous oxide from soil. *Current Science* 77(3):359-369.
- Paul, E.A., and F.E. Clark. 1989. *Soil Microbiology and Biochemistry*. Academic press, San Diego.
- Paustian, K., H.P. Collins, and E.A. Paul. 1997. Management controls in soil carbon. *In* Paul E.A., E.T. Elliot, and C.V. Cole (eds). *Soil organic matter in temperate agroecosystems. Long term experiments in North America*. CRC Press, Boca Raton, FL.
- Pomes, M.L, D.H. Wilkinson, and P.B. McMahon. 1998. Nitrous oxide fluxes from a clay pan soil overlying nitrate-enriched glacial drift. *J. Env. Health* 6:1-14.
- Quemada, M., and M.L. Cabrera. 1997. Temperature and moisture effects on C and N mineralization from surface applied crimson clover residue. *Plant and Soil* 189:127-137.
- Reicosky, D.C. 1991. Effects of conservation tillage on soil organic carbon dynamics: Field experiments in the U.S. corn belt. *In*: D.E Stott, R.H. and G.C. Stein (eds). *Sustaining the global farm. Selected papers from the 10th International Soil Conservation Organization meeting held May 24-29 at Purdue University and the USDA National Soil Erosion Research Laboratory*.
- Reicosky, D.C. 1997. Tillage-induced CO₂ emission from soil. *Nutr Cycling Agroecosyst.* 49:273-285.
- Reicosky, D.C. and D.W. Archer. 2007. Moldboard plow tillage depth and short-term carbon dioxide release. *Soil Tillage Res.* 94(1):109-121.
- Rochette, P., D.A. Angers, M. H. Chantigny, N. Bertrand and D. Cote. 2004a. Carbon dioxide and nitrous oxide emissions following fall and spring applications of pig slurry to an agricultural soil. *Soil Sci. Soc. Am. J.* 68:1410-1420.
- Rochette, P., R.R Simard, N. Ziadi, M.C. Nolin, and A.N. Cambouris. 2004b. Atmospheric composition and N₂O emissions in soils of contrasting textures fertilized with anhydrous ammonia. *Can. J. Soil. Sci.* 84:339-352.

- Rochette, P., D.A. Angers, M.H. Chantigny, N. Bertrand and D. Cote. 2006. In situ mineralization of dairy cattle manures as determined using soil-surface carbon dioxide fluxes. *Soil Sci. Soc. Am. J.* 70:744-752.
- Rozas, H.R., H.E. Echeverria, and L.I. Picone. 2001. Denitrification in maize under no-tillage: effect of nitrogen rate and application time. *Soil Sci. Soc. Am. J.* 65:1314-1323.
- Ruffo, M.L and G.A. Bollero. 2003. Residue decomposition and prediction of carbon and nitrogen release rates based on chemical fractions using principal-component regression. *Agron J.* 95:1034-1040.
- Ruffo, M.L., and G.A. Bollero. 2003. Modeling rye and hairy vetch residue decomposition as a function of degree-days and decomposition days. *Agron. J.* 95:900-907.
- Sainju, U.M., J.D. Jabro and W.B. Stevens. 2008. Soil carbon dioxide emission and carbon contents as affected by irrigation, tillage, cropping system, and nitrogen fertilization. *J. Environ. Qual.* 37:98-106.
- Salas, A.M., E.T. Elliot, D.G. Westfall, C.V. Cole and J. Six. 2003. The role of particulate organic matter in phosphorus cycling. *Soil Sci. Soc. Am. J.* 67:181-189.
- Sarrantonio, M. 2003. Soil response to surface-applied residues of varying carbon-nitrogen ratios. *Biol. Fertil. Soils* 37:175-182.
- Schaffers, A.P., M.C. Vesseur and K.V. Sykora. 1998. Effects of delayed hay removal on the nutrient balance of roadside plant communities. *J. Appl. Ecology* 35:349-364.
- Schmid, M., J. Fuhrer, and A. Neftel. 2001. Nitrous oxide concentrations in the soil of a mown grassland: comparison of model results with soil profile measurements. *Water Air Soil Pollut. Focus.* 1:437-446.
- Seghers, D., E.M. Top, D. Rehuel, R. Bulcke, P. Boeckx, W. Verstraet, and S.D. Siciliano. 2003. Long-term effects of mineral versus organic fertilizers on activity and structure of the methanotrophic community in agricultural soil. *Environ. Microbiol.* 5(10):867-877.

- Sehy, U., R. Russer and J.C. Munch. 2003. Nitrous oxide fluxes from maize fields: relationship to yield, site-specific fertilization, and soil conditions. *Agric. Ecosyst. Environ.* 99(1-3):97-111.
- Six, J., S.M. Ogle, F.J. Breidt, R.T. Conant, A.R. Mosier and K. Paustin. 2004. The potential to mitigate global warming with no-tillage management is only realized when practiced in the long term. *Glob. Change Biol.* 10:155-160.
- So, H.B., R.C. Dalal, K.Y. Chan, N.M. Menzies, and D. M. Freebairn. 2001. Potential of conservation tillage to reduce carbon dioxide emission from Australian soils. p.821-826. *In* D.E Scott, R.H. Mohtar and G.C. Steinhardt (eds). *Sustaining the Global Farm. Selected paper for the 10th International Soil Conservation Organization meeting held May 24-29, 1999 at Purdue University and the USDA-ARS National Soil Erosion Research Laboratory.*
- Stanford, G. and S.J. Smith. 1972. Nitrogen mineralization potentials of soils. *Soil Sci. Soc. Am. J.* 36:465-472.
- Suwanwaree, P. and G.P. Robertson. 2005. Methane oxidation in forest, succession, and no-till agricultural ecosystems: effects of nitrogen and soil disturbance. *Soil Soc. Am. J.* 69:1722-1729.
- Swift, M.J., O.W. Heal, and M.J. Anderson (eds). 1979. *Decomposition in terrestrial ecosystems.* Blackwell Scientific, Oxford.
- Terra, J.A., J.N. Shaw, D.W. Reeves, R.L. Raper, E. van Santen, E.B Schwab, and P.L. Mask. 2006. Soil management and landscape variability affects field-scale cotton productivity. *Soil Sci. Am. J.* 70:98-107.
- Thonissen, C, D.J. Midmore, J.K. Ladha, D.C. Olk and U. Schmidhalter. 2000. Legume decomposition and nitrogen release when applied as green manures to tropical vegetable production systems. *Agron. J.* 92:253-260.
- Trinsoutrot, I., S. Raceous, B. Bentz, M. Lineras, D. Cheneby and B. Nicokardit. 2000. Biochemical quality of crop residues and carbon and nitrogen mineralization kinetics under no limiting nitrogen conditions. *Soil Sci. Soc. Am. J.* 64:918-926.
- Venterea, R.T., M. Burger and K.A. Spokas. 2005. Nitrogen oxide and methane emission under varying tillage and fertilizer management. *J. Environ. Qual.* 34:1467-1477.

- West, T.O and G. Marland. 2002. A synthesis of carbon sequestration, carbon emissions, and net carbon flux in agriculture: comparing tillage practices in the United States. *Agric., Ecosyst. Environ.* 91:217-232.
- West, T.O, and G. Marland. 2003. Net carbon flux from agriculture: carbon emission, carbon sequestration, crop yield and land use change. *Biochemistry.* 11:73-83.
- White, T.A., D.J. Baker and K.J. Moore. 2004. Vegetation diversity, growth, quality and decomposition in managed grasslands. *Agric. Ecosyst. Environ* 101:73-84.
- Willison, T.W., C.P. Webster, K.W.T. Goulding, and D.S. Powlson. 1995. Methane oxidation in temperate soils: effect of land use and the chemical form of nitrogen fertilizer. *Chemosphere.* 30(3):539-546.
- Wood, C.W., D.G. Westfall, G.A. Peterson, and I.C. Burke. 1990. Impacts of cropping intensity on carbon and nitrogen mineralization under no-till dryland agroecosystems. *Agron. J.* 82: 1112-1120.

APPENDICES

Appendix 1. Terrain attributes at E.V. Smith Research Center site, near Shorter, AL. Data is mean of each soil landscape position

Landscape	Elevation (m)	Slope (%)	CT†	PLAN‡	PROF§	FA§¶	SHWT#§	sand	silt	clay
Summit	71	0.81	3.98	0.01	0.02	5.10	146	57	24	19
Drainageway	69	1.18	6.16	-0.08	-0.09	30.35	75	64	25	11
Sideslope	71	3.40	4.15	-0.01	0.02	7.13	108	54	24	21

† Compound topographic index

‡ Planimetric curvature

§ Profile curvature

¶ Flow accumulation

Seasonal high water table

Appendix 2. Analysis of variance of gas fluxes among summit, drainageway and sideslope landscape positions at E.V. Smith Research Center near Shorter, AL.

Source of variation	CH ₄ †	N ₂ O	CO ₂	Soil organic C	Total soil N	NH ₄	NO ₃	Gravimetric soil moisture
-----P-value-----								
Soil management (M)	0.312	0.212	0.368	0.001	0.001	0.072	0.001	0.001
Landscape position (L)	0.495	0.004	0.469	0.328	0.048	0.002	0.102	0.035
Season (S)	-	0.001	0.001	0.001	0.001	0.001	0.001	0.001
M x L	0.346	0.037	0.722	0.964	0.049	0.192	0.152	0.675
M x S	-	0.031	0.001	0.001	0.129	0.001	0.0001	0.001
L x S	-	0.002	0.684	0.235	0.417	0.001	0.293	0.003
L x M x S	-	0.224	0.154	0.428	0.636	0.001	0.166	0.417

† CH₄ data is from one season (spring 2004) only

Appendix 3. Pearson correlation coefficients (r) relating soil organic C and total N with terrain attributes after six years of tillage and dairy manure application.

		Soil organic C				Soil Total N			
		CT	CTM	CsT	CsTM	CT	CTM	CsT	CsTM
Elevation (m)	r	0.404	0.091	0.009	0.128	0.604	-0.047	0.216	0.028
	P	0.281	0.815	0.982	0.742	0.085	0.904	0.576	0.943
Slope (%)	r	-0.293	-0.388	0.011	-0.627	-0.159	-0.035	-0.102	-0.598
	P	0.444	0.303	0.978	0.071	0.682	0.929	0.794	0.089
CTI†	r	0.036	0.102	0.140	0.257	-0.182	-0.227	-0.066	0.342
	P	0.927	0.795	0.720	0.504	0.640	0.558	0.866	0.368
Planimetric curvature	r	0.319	-0.181	-0.082	0.114	0.312	-0.324	0.218	0.049
	P	0.403	0.640	0.834	0.770	0.414	0.395	0.573	0.900
Profile curvature	r	0.267	-0.153	-0.355	0.082	-0.004	-0.271	-0.323	-0.008
	P	0.488	0.694	0.349	0.834	0.991	0.480	0.396	0.984
FA‡	r	0.410	0.640	-0.093	-0.239	-0.018	0.315	-0.261	-0.147
	P	0.273	0.064	0.811	0.535	0.963	0.409	0.497	0.707
SHWT (cm) §	r	0.392	0.147	0.120	-0.074	0.602	-0.186	0.385	-0.041
	P	0.296	0.706	0.759	0.850	0.086	0.632	0.307	0.916
Sand (%)	r	0.036	0.173	0.469	0.162	-0.345	0.031	0.308	0.146
	P	0.926	0.657	0.203	0.678	0.363	0.937	0.420	0.709
Silt (%)	r	0.200	-0.349	-0.378	0.002	0.308	-0.494	-0.223	0.028
	P	0.606	0.357	0.316	0.996	0.420	0.177	0.564	0.943
Clay (%)	r	-0.074	-0.101	-0.237	-0.186	0.328	0.060	-0.170	-0.190
	P	0.850	0.796	0.539	0.632	0.388	0.879	0.663	0.625

† Compound topographic index

‡ Flow accumulation

§ Seasonal high water table

Appendix 4. Analysis of variance of soil C and N mineralization among summit, drainageway and sideslope landscape positions at E.V. Smith Research Center near Shorter, AL.

Source of variation	Soil Organic C	Soil Total N	Cumulative C mineralized	Cumulative N mineralized	Relative N mineralized	C turnover	C:N mineralized
-----P-value-----							
Soil management (M)	0.001	0.001	0.003	0.001	0.001	0.592	0.820
Landscape position (L)	0.190	0.364	0.618	0.244	0.124	0.484	0.502
M x L	0.482	0.590	0.090	0.872	0.707	0.793	0.640

Appendix 5. Analysis of variance of cover crop mass C, N and K rate constants (*k*) at E.V. Smith Research Center near Shorter, AL.

Source of variation	Mass <i>k</i>	C <i>k</i>	N <i>k</i>	K <i>k</i>
	-----P-value-----			
Crop residue (R)	0.001	0.024	0.001	0.001
Placement method (P)	0.001	0.001	0.001	0.002
R x P	0.001	0.001	0.002	0.012

Appendix 6. Analysis of variance of P mineralization among summit, drainageway and sideslope landscape positions at E.V. Smith Research Center near Shorter, AL.

Source of variation	P mineralization
Crop residue (R)	0.002
Placement method (P)	0.001
Landscape position (L)	0.001
R x L	0.047
R x P	0.001
R x L x P	0.055