

Carbon and Nitrogen Cycling under Warm Season Turfgrasses

by

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Abstract

Despite its large-scale presence in the urban ecosystem, the role of turfgrasses in C and N cycling in urban soils in the southeastern United States has not been documented, and in particular C and N cycling studies in warm-season turfgrasses are lacking. The objectives of the proposed study were: 1) determine C sequestration under three major warm-season turfgrass species including: bermudagrass (*Cynodon dactylon* (L.) Pers. x *C. transvaalensis* Burt-Davy), centipedegrass (*Erecholmoa ophroides* Munroe Hack.), and zoysiagrass (*Zoysia japonica* Stued.), 2) determine CO₂ flux from soil as affected by N applied to bermudagrass, and, 3) determine decomposition rates and C and N release of warm and cool season turfgrass clippings.

The first objective was initiated in the winter of 2012 and conducted for two years on a Marvyn loamy sand (fine-loamy, kaolinitic, thermic Typic Kanhapludult). Eighteen lawns were sampled twice per year, six lawns of each grass species, with the harvested grasses separated into stems (rhizomes and/or stolons), above ground biomass (verdure), belowground roots, and belowground soil. Results showed an accumulation of organic C in the top 20 cm of the soil profile, with C sequestration ranging from 2.3±0.2 (bermudagrass) to 4.3±0.4 (zoysiagrass) Mg ha⁻¹ yr⁻¹.

The second objective was initiated in March, 2012 on eight-year-old 'Tifway' hybrid bermudagrass plots on a Marvyn loamy sand. The experimental design was a randomized complete block with four N rates of 24, 49, 98, and 196 kg N ha⁻¹ yr⁻¹,

replicated three times. Carbon dioxide flux was measured weekly for 95 weeks using an automated soil CO₂ flux system. Results showed strong correlation between CO₂ flux and N rate, and CO₂ flux during the study period significantly increased from 107±4.5 to 144±4.4 Mg ha⁻¹ as N rate increased from 24 to 196 kg N ha⁻¹.

The third objective was initiated on May 17, 2012 and conducted for 46 weeks at the Auburn University Turfgrass Research Unit. Litter from five turfgrasses was selected for this study including: bermudagrass, centipedegrass, St. Augustinegrass (*Stenotaphrum secundatum* L.), tall fescue (*Lolium arundinaceum* S.J. Darbyshire), and zoysiagrass. Litter was placed into nylon bags measuring 10 × 20 cm with 50 to 60 μm openings based on an oven dry rate of 3.6 Mg ha⁻¹. Litter bags were retrieved from the field after 0, 1, 2, 4, 8, 16, 24, 32, and 46 weeks, and retrieved bags were analyzed for total C and N. A double, four-parameter exponential decay model was used to describe mass, C, and N loss. Results indicated that tall fescue decomposition occurred more rapidly compared to other turfgrasses. In all, established turfgrasses are significant C sequestrators, although the degree of this C storage is affected by grass species and N fertilization.

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List of Abbreviations

ADF	Acid detergent fiber
ADL	Acid detergent lignin
AFDW	Ash free dry weight
AL	Alabama
ANOVA	Analysis of variance
BD	Bulk Density
C	Carbon
CEC	Cation exchange capacity
CH ₄	Methane
CO ₂	Carbon dioxide
cm	Centimeter
d	day
D	Soil depth
Diam	Diameter
g	Gram
G	Giga
GHG	Greenhouse gas
H ₂ O	Water vapor
ha	Hectare

hr	Hour
HS	humic substances
IPCC	Intergovernmental Panel on Climate Change
kg	Kilogram
m	Meter
Mg	Megagram
mm	Millimeter
mo	month
N	Nitrogen
N ₂ O	Nitrous oxide
NCDC	National Climate Data Center
NDF	Neutral Detergent Fiber
NOAA	National Oceanic and Atmospheric Administration
NRC	National Research Council
NRCS	Natural Resources Conservation Service
O ₃	Ozone
OM	Organic Matter
ppmv	parts per million by volume
T	Ton
TGRU	Turfgrass Research Unit
USDA	United States Department of Agriculture
USEPA	United States Environmental Protection Agency
wk	week

yr

year

CHAPTER I

Introduction and Literature Review

Industrialization and urbanization have altered atmospheric composition, and increased greenhouse gas (GHG) concentrations have caused global climate change. Greenhouse gases include carbon dioxide (CO₂), water vapor (H₂O) (Ledley, et al., 1999; Matthews, 1996), ozone (O₃) (Ledley, et. al., 1999), methane (CH₄), and nitrous oxide (N₂O) (Ledley, et al., 1999; Morrissey and Justus, 1997). Atmospheric CO₂ contributes about 25% to the greenhouse effect, while H₂O, O₃, and CH₄ plus N₂O contribute $\approx 60\%$, $\approx 8\%$, and $\approx 7\%$, respectively (Karl and Trenberth, 2003).

Atmospheric CO₂ concentration has increased by $\approx 42\%$ from 280 ppmv in 1850 to 398 ppmv in 2014 (NOAA, 2014). This increase, along with the probability of increasing global temperatures (Rustad et al, 2000), is expected to alter the distribution of carbon (C) between the atmosphere, vegetation and soils (Watson et al., 2000). In 2005, CO₂ represented more than 54% of total greenhouse gas emissions (NRC, 2011). Climatologists estimate that each 1000 Gt of C (≈ 3652 Gt CO₂) emissions increases global temperature on average by ≈ 2 °C (NRC, 2011).

Average global temperatures have increased approximately 0.6 ± 0.2 °C since 1750, and 12 (2001-2012) out of the 14 warmest years on record have occurred since 1880 (NOAA-NCDC, 2012). Furthermore, climatologists expect that average global

temperatures could increase by 2.5 ± 0.6 °C in the next fifty years (NRC, 2011). In addition to a sea level rise of 20 cm since 1870 (IPCC, 2007), there have been remarkable changes in the ocean's biology, permafrost (NRC, 2011), ecosystems (Greene and Pershing, 2007) and wildfire occurrence and extent (Running, 2006; NRC, 2011). These changes are caused by additional greenhouse gas emissions originating from human activities, including land use change, soil cultivation, biomass burning and fossil fuel combustion (NRC, 2011). Schrag (2007) suggested that reducing temperature effects requires sequestration of C in soil.

The role of Soil in the C cycle

Soils are a major component of the global C cycle (Lal et al., 2004). Carbon is mostly present in the atmosphere as CO₂, with minor amounts of CH₄, CO and other gases. In fact, CO₂ does not react with other gases in the atmosphere. Its atmospheric concentration is regulated by interactions with the soil surface during annual cycles of photosynthesis, respiration, and gas exchange (Schlesinger, 1997).

Understanding the role of ecosystems and underlying soils as a sink or a source for C on a global scale is necessary for estimating changes in atmospheric CO₂ concentration (Johnson and Curtis, 2001). Any disruption of soil due to land use or climate change can alter soil C pools and result in significant impacts on the C budget of the atmosphere. In contrast, increasing C storage in the world's soils may reduce concentrations of atmospheric CO₂ (Rustad et al., 2000). Appropriate land use and soil management play major roles in reducing the greenhouse effect by increasing C stored within terrestrial ecosystems (Lal, 1997). For example, rates of soil organic carbon (SOC)

sequestration under a subtropical wet forest plantation were higher than under a cool temperate-zone pine plantation, with an average accumulation of $33.8 \text{ g C m}^{-2} \text{ yr}^{-1}$ (Post and Kwon, 2000). Perennial grasses are more effective than woody plants at sequestering C in soils (Post and Kwon, 2000). Another example indicates significant changes in C sequestration where sugarcane fields were converted to fast growing eucalyptus plantations (Bashkin and Binkley, 1998). After 10-15 yr, SOC increased under eucalyptus in Hawaii by an average of 19.4 Mg ha^{-1} in the top 55 cm of soil. About 49% of the world's organic C stocks are stored in boreal forests compared to 37 and 14% in tropical and temperate forests, respectively (Post et al. 1982). It is clear that the amount of organic C stored in soils varies with ecosystems, land use and management practices (Jobbagy and Jackson, 2000).

CO₂ Capture, Storage, and Soil Respiration

The idea behind C sequestration is to find large reservoirs to store CO₂ instead of allowing it to discharge into the atmosphere, reducing the environmental alteration resulting from increasing temperatures, and/or recycling of captured CO₂. For example, after capturing and separating CO₂ from combustion stack gases, liquid CO₂ can be transported and discharged into the bottom of the ocean, stored in geological formations, stored in a dry ice form (Matthews, 1996), or converted to valuable materials through biological or chemical processes (Beecy, et al., 2001).

Carbon sequestration occurs through direct and indirect fixation of atmospheric CO₂ into various reservoirs such as geological, oceans, plant biomass, and soils. Soils are the second largest C reservoir after the deep ocean (Schimel et al., 1994) with a pool of

2500 Gt (Lal et al., 2004). Carbon sequestration is estimated to occur from 10 to 20 years following management change, after which the soil approaches a new C equilibrium (Kern and Johnson, 1993).

The amount of C a particular soil can sequester varies and is dependent on climate, topography, and the ecosystem. A short summer season and cool mean annual temperatures are less favorable for microbial activity. An analysis of long-term Canadian experiments included forest and agricultural lands showed that the amount of SOC that could be sequestered during 25 to 30 years was estimated to be 50 to 75 g C m⁻² yr⁻¹ (Dumanski et al., 1998).

Plants fix atmospheric CO₂ via photosynthesis and turn it into biomass. This biomass enters the soil through decomposition of aboveground biomass or belowground from roots and their exudates that are subsequently transferred to soil microbes and turned into SOC through decomposition processes. Some of this C is released back as CO₂ via soil respiration.

Soil respiration is a composite flux that includes respiration of soil organisms and plant roots (Doff et al., 2004; Subke et al., 2006). Root respiration contributes more CO₂ to total soil respiration than all other processes (Hanson et al., 2000) with estimated values between 30 and 70% in temperate forests (Buchmann, 2000), 50 to 93% in tundra ecosystems (Raich and Tufekcioglu, 2000), and 17 to 60% in grasslands (Kucera and Kirkham, 1971; Coleman, 1973; Herman, 1977; Buyanovsky et al., 1987; Dugas et al., 1999).

Temperature is the most important environmental factor affecting soil respiration (Kirschbaum, 1995; Davidson et al., 1998), and soil respiration increases with increasing

temperature (Raich and Schlesinger, 1992; Rustad and Fernandez, 1998). However, components of soil respiration exhibit different sensitivities to temperature (Kowalenko et al., 1978). Due to the effect of temperature on microbial respiration, decomposition rate of the labile fraction of organic matter is also controlled by temperature (Giardina and Ryan, 2000; Lenton and Huntingford, 2003).

The temperature sensitivity of soil organic matter (SOM) decomposition is commonly presented as a Q_{10} (Rustad et al., 2000). The Q_{10} for a reaction rate is defined as the factor by which the rate increases with a 10°C rise in temperature (Parkin and Kaspar, 2003; Davidson and Janssens, 2006). The Q_{10} for CO₂ evolution varies with soil temperature (Liefeld and Fuhrer, 2005; Kirschbaum, 2006) and soil depth (Swanson and Flanagan, 2001). Various studies reported different Q_{10} for CO₂ evolution under different soil conditions over different temperature with ranges from 1.55 to 2.70 (Howard and Howard, 1993; Lloyd and Taylor, 1994).

Soil warming has a greater effect on the rate of CO₂ emitted from soils in the boreal and tundra region than in temperate and tropical regions (Niklinska et al., 1999), due to greater concentrations of labile soil C in the boreal and tundra region (Schlesinger and Andrews, 2000). Soil respiration accounts for about 25% of the C exchange between the biosphere and the atmosphere (Raich and Schlesinger, 1992). Soil respiration in soils developed under tropical lowland forests was estimated to be 1092 g m⁻² yr⁻¹ compared to 662, 544, and 200 g m⁻² yr⁻¹ for soils developed under temperate forests, cultivated land, and desert scrub, respectively (Raich and Schlesinger, 1992). Globally, soil respiration is estimated to range from 68 x 10¹⁵ to 77x 10¹⁵ g C yr⁻¹ (Schlesinger and Andrews, 2000).

Soil moisture content also influences soil respiration, and dry soils typically respire less than their wet counterparts (Davidson et al., 2000; Pangle and Seiler, 2002; Reichstein et al., 2002). Low soil moisture suppresses soil microbial populations (Schimel et al., 1999) and their activities, limiting soil respiration. Rochette et al. (1990) found that soil respiration increased by 90% following a rainfall preceded by a long dry period. Davidson et al. (2000) measured soil respiration in a primary forest ($2.0 \text{ kg C m}^{-2} \text{ yr}^{-1}$), a secondary forest ($1.8 \text{ kg C m}^{-2} \text{ yr}^{-1}$), an active cattle pasture ($1.5 \text{ kg C m}^{-2} \text{ yr}^{-1}$) and a degraded cattle range ($1.0 \text{ kg C m}^{-2} \text{ yr}^{-1}$) in eastern Amazonia. They found that soil respiration decreased from the wet to the dry season in all soils under study.

Besides soil temperature and moisture, soil respiration may be affected by other factors such as root biomass and microbial population (Buchmann, 2000; Li et al., 2004; Dilustro et al., 2005). Li et al. (2004) measured CO_2 efflux and microbial biomass at a soil depth of 25 cm in a secondary forest dominated by *Tabebuia heterophylla* and a pine (*Pinus caribaea*) plantation in the Luquillo Experimental Forest of Puerto Rico. In that study four treatments were included: 1) root exclusion, 2) litter exclusion, 3) mixed root and litter exclusion, and, 4) control plots without litter or root exclusion. Soil CO_2 efflux decreased significantly under control plots in the pine plantation and the secondary forest, while litter exclusion plots had a greater effect on soil CO_2 efflux in the plantation than the secondary forest. Soil CO_2 efflux was decreased from 2.33 ± 0.12 to $0.75 \pm 0.06 \text{ g C m}^{-2} \text{ d}^{-1}$ and from 2.66 ± 0.16 to $1.21 \pm 0.11 \text{ g C m}^{-2} \text{ d}^{-1}$ under litter exclusion in the plantation and the secondary forest, respectively. Conversely, total soil microbial biomass was lowest under the mixed litter and root exclusion plots in the plantation and secondary forest. Total microbial biomass in root exclusion plots was lower in the plantation soil

(711 ± 43 to 489 ± 22 mg C k^{-1} soil) compared to the secondary forest soil (618 ± 45 to 256 ± 28 mg k^{-1} soil) (Li et al., 2004).

Soil Organic Matter (SOM)

Soils play an important role in the formation, maintenance and turnover of organic matter. Soil organic matter determines many of the biological, physical and chemical properties of soils (Merino et al., 2004; Grace et al., 2006). Accumulation and storage of OM in soil depends on the addition of fauna, flora and/or loss of C from soil through respiration and physical activities (Baldock and Skjemstad, 2000).

The amount of SOM in soils is a function of climate, parent material, topography, time and human activities (Jenny, 1941; Schimel et al., 1994; Lal et al., 1998 a,b; Rice, 2005). These factors affect vegetation cover and decomposition processes. Climate, through soil moisture and temperature regimes, plays an important role in plant productivity and decomposition rates. Parent material affects soil mineralogy, texture, structure, and soil pH, all factors that influence the formation of stable soil aggregates. Increasing soil C inputs over time contributes to a gradual buildup of SOM. Human activities in terms of land use, crop rotation management, and tillage systems that increase C input will increase SOM. Tillage systems that disturb the soil usually result in a reduction of SOM via exposure of SOM which stimulates microbial degradation and respiration.

Land management practices have a profound effect on SOM. Traditional U.S. agriculture has included tillage, which is primarily used in crop seedbed preparation and weed control. Tillage practices and land use intensification have led to substantial

reductions in SOM. A better understanding of agro-ecosystem dynamics has also led to the identification of alternative practices that facilitate soil preservation. Three basic guidelines have been identified to maximize SOM if cropping must be continued: (1) minimize soil disturbance and erosion, (2) retain crop residues within the soil, and, (3) increase water and nutrient efficiencies within the cropping system (Paustian et al., 1997).

Soil Organic Carbon (SOC)

Soil OC is estimated to make up two-thirds of terrestrial C (Post et al., 1990; Schlesinger, 1995). Soil OC is the main component of soil organic matter (SOM). Each molecule of SOM contains 58% C on average; however, organic C content can vary considerably with molecular composition (Stevenson and Cole, 1999). Three primary categories of organic materials found in the soil include labile, intermediate, and recalcitrant SOM (Wander, 2004).

The active pool consists of substances such as carbohydrates that are easily broken down via a vast range of microorganisms with turnover times estimated to be from months to a few years (Derenne and Largeau, 2001). The slow pool, represents stabilized decomposition products such as lignin with turnover times ranging from decades to hundreds of years (Coleman and Jenkinson, 1996). The passive pool is recalcitrant SOC and includes products such as coal with residence times of 5000 to 10000 yr (Skjemstad et al., 2002; Swift, 2001). In the slow and recalcitrant pools, SOC is stabilized due to physical protection and chemical recalcitrance against microbial degradation. Physical protection is provided by physico-chemical interaction between the

soil minerals and organic material. Physical stabilization is due to aggregate formation. These mechanisms are not independent but interact with each other (Krull et al., 2003). The aggregate provides physical protection to the SOC unless disrupted and exposed to factors that promote microbial degradation. Krull et al., (2003) suggested that chemical recalcitrance may be the only mechanism for SOC protection over long time periods.

Chemical recalcitrance is achieved during the process of humification (Balsler, 2005), which is a part of decomposition that leads to formation of stable humic substances (HS). In addition, HS are important in determination of soil properties such as water holding capacity, soil structure, aggregation, and nutrient retention. Humic substance formation and turnover in soil is governed by microorganisms (Balsler, 2005). Humification plays an important role in the process of SOC sequestration. The chemical perspective on the role of microbes in humus formation suggests the following pathways of HS formation (Balsler, 2005): (1) degradation of plant material and abiotic condensation of the products to form humic substances, (2) degradation of primary resources and re-synthesis into recalcitrant components in the bodies of microbes which remain in the soil after their death, (3) selective preservation wherein easily degradable plant material is degraded and the large resistant structures form humic substances, and (4) microbes participate directly in humus formation by enzymatic activities. Therefore, from a microbiological perspective, humification occurs in three phases (Balsler, 2005): First is a rapid initial decomposition of primary plant residues which is performed by a wide range of microorganisms; Second is a slow decomposition of primary plant structural components; in this step the degradation rate slows because labile C has been used by microorganisms, and larger plant structures such as lignin, cellulose and

hemicelluloses remain. This plant structure are larger in size than labile C from phase 1 and must be decomposed outside the cell prior to uptake into the cell for degradation; and lastly is an alteration of SOC and HS genesis - in this phase microbial activity oxidizes and alters HS and/or generates and polymerizes aromatic compounds to form HS. Since humification is controlled by soil microbes, the same factors affecting soil microbes also affect the process of HS formation. In case of climatic conditions that do not favor the growth of microorganisms the plant residue will be stored in the soil as SOC (Balsler, 2005).

In the tundra ecosystem, low temperatures limit the microbial utilization of organic matter, which leads to an accumulation of SOC (Archibold, 1995). But this C is in an unstable state. Increases in temperature due to global climate change in this ecosystem may result in microbial decomposition and loss of SOC (Balsler, 2005).

The composition of plant residues, in particular C, N, and lignin concentrations, determines the rate and extent of decomposition of such residues. Turfgrasses produce thatch, which is a layer of intermingled living and dead plant material that underlies the green turf canopy and overlies the soil, which has a high lignin concentration that retards degradation of cellulosic cells (Ledeboer and Skogley, 1967). Raturi et al. (2004) found that thatch has a high C concentration that increases C/N ratio, thus slowing decomposition. With slow decomposition owing to high C/N ratio and lignin concentrations in turf plant residues, retention of C in turf systems increases with time until a quasi-equilibrium is established within the bounds of soil and climatic controls.

The SOC can be protected as long as there is no soil disturbance. Once a disturbance occurs SOC is exposed to microbial degradation, which may or may not lead

to HS formation depending on the structure of the microbial community. For example, fungi can degrade plant residue more efficiency than bacteria because fungal cell walls are highly recalcitrant, and release more C as CO₂ (Balsler, 2005).

Another part of the remaining C is leached out as dissolved organic C (DOC). According to Hartikainen and Yli-Halla (1996) the use of mineral fertilizers can increase the solubility of SOM, thus increasing soil DOC content. This DOC can leach to lower soil depths and remain protected from further degradation and/or loss as CO₂. This is an important process of SOC sequestration known as illuviation (Krull et al., 2003).

Turfgrass Impacts on C sequestration

Milesi et al. (2005) estimated the total U.S. turfgrass area to be 163,800 ± 35,850 km². Turfgrass cover in the U.S. is rapidly expanding owing to increasing urbanization and addition of approximately 700,000 ha of residential property each year (Robbins and Birkenholtz, 2003). Such soil covers could have a significant effect on soil C sequestration. Soils developed under temperate grassland sequestered a C mass of 192 Mg ha⁻¹ compared to 127, 118, and 56.1 Mg ha⁻¹ for soils developed under cultivated agriculture, temperate forest, and desert, respectively.

Turfgrasses are a unique management scenario where soils are rarely inverted, there is no soil disturbance due to plowing, a long-term perennial crop provides constant cover, and harvest via removal of clippings is performed frequently. In many cases turfgrasses are also intensively managed with frequent fertilization, irrigation, and application of pesticides.

Synthesis of historic soil testing data from golf courses in Denver and Fort Collins, CO, indicated that SOC sequestration in golf course turf soils occurs at a high rate of $1 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ comparable to that under Conservation Reserve Program land (Qian and Follett, 2002). In addition, turfgrasses produce thatch. It has been shown that thatch is a temporary C sink (Raturi et al., 2005), and that N in the thatch can affect C and N cycling. Total, soluble and lignin C were significantly greater in the thatch of both cool (*Agrostis palustris* Huds.) and warm (*Zoysia japonica* Steud.) season turfgrasses than that measured in the underlying soil (Raturi et al., 2005).

A study of urban green space used two residential blocks in the Chicago area as the study site, with green space defined as ‘any soil surface capable of supporting vegetation’ (Jo and McPherson, 1995). Collected data included foliage, root and total biomass of numerous tree species, calculated using previously published biomass equations. Clippings were collected biweekly from 26 lawns, from Oct 1992 through September 1993, except in winter when mowing was not needed. Carbon storage in grass followed a predictable pattern, with maximum C in summer (July) followed by a decline in fall and winter (Jo and McPherson, 1995). Calculated turnover rates for stubble and root C were 2 and 2.9 years, respectively, indicating that about 50% of the stubble C and 34% of the total root C would be replaced each year. Average C concentration of the grass was 42.7% on a dry weight basis, with no difference in C concentration between live root, stubble or clippings. Over the year there was a net C release from turfgrass maintenance, with the greatest C release coming from mowing. Annual C released from mowing, irrigation and fertilization was 0.14 kg m^{-2} , while soil (0-60 cm) C storage was 20.6 kg m^{-2} , and organic C represented 79% of all stored C. Trees and shrubs represented

10.8 to 10.5% of that C, while the turfgrass presented 0.6%. In this study, because of mowing, grass returned 1.5 times as much C to the atmosphere as was sequestered. However, the studied green spaces were net sinks for C, owing to storage in soils and woody plants (Jo and McPherson, 1995).

In Colorado, C fluxes in Kentucky bluegrass lawns were compared to those in irrigated corn (*Zea mays* L.), wheat (*Triticum aestivum* L.) fallow, and native grasslands (Kaye et al., 2005). In that study, organic soil C contents were determined in a June sampling and potential C mineralization was measured using a laboratory incubation procedure. Soil respiration was measured twice a month, and C and N content of all plant tissues were also measured during the study. Harvested turfgrass clipping samples were analyzed for N, while stubble biomass turnover was estimated using previously published Kentucky bluegrass data (Falk, 1976; 1980; Jo and McPhearson, 1995). Clippings were returned to the lawn. Compared to the other studied land-uses, total belowground C allocation in the urban lawn system was 2.5 to 5 times greater ($2,602 \text{ g C m}^{-2} \text{ yr}^{-1}$) (Kaye et al., 2005). Seasonal variation in C from grass clippings was similar to that observed in Jo and McPhearson (1995), with a high in May ($43 \text{ g C m}^{-2} \text{ mo}^{-1}$), and a steady decline thereafter. The authors concluded that urbanization of arid and semiarid ecosystems leads to enhanced C cycling, when compared to native grasslands and agricultural ecosystems (Kaye et al., 2005).

Qian et al. (2010) evaluated C sequestration under different turfgrass species. The cool season grasses fine fescue (both irrigated and non-irrigated), Kentucky bluegrass (irrigated), and creeping bentgrass (irrigated) were evaluated for changes in soil organic C, soil C sequestration and soil organic C decomposition. Four years after establishment

approximately 17 to 24% of soil organic C (0 - 10 cm depth) was contributed by the established turfgrass. The amount of soil organic C increase differed with turfgrass species and irrigation, with irrigated fine fescue having the highest soil organic C, which was estimated to be $3.4 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$. All turfgrass species sequestered C during the first four years after turf establishment, with the highest rates for irrigated fine fescue and creeping bentgrass (Qian et al., 2010).

Allaire et al. (2008) measured CO_2 emissions from four different levels of lawn management. Four lawn management approaches used on the mixed cool-season lawn were: 1) N fertilizer with clipping removal and frequent mowing, 2) no fertilizer with clippings returned and frequent mowing, 3) treatment #2 with only 3 mowing, and, 4) treatment #2 with only 1 mowing. Carbon dioxide emission data was collected every week for one year. Mowing frequency had a greater impact on CO_2 flux than fertilizer. Frequently mowed lawns had an annual CO_2 emission of $2.0 \text{ kg m}^{-2} \text{ yr}^{-1}$, which was four times higher than in lawns mowed 3 or 1 times. At a high mowing frequency clipping removal did not affect CO_2 emissions (Allaire et al., 2008).

Long-term soil test data and the CENTURY model were utilized in a series of papers that assessed soil C sequestration in golf courses located in the Denver and Fort Collins, CO areas (Qian and Follett, 2002). Courses ranged in age from 2 to 45 years, with prior use being native grassland or agriculture. Using nonlinear regression analysis it was determined that the time required for soil organic matter to reach equilibrium for fairways and putting greens was 31 and 45 years, respectively. Total C sequestration in fairways during the first 25 to 30 years after turf establishment was estimated to be 0.9 t

ha⁻¹ yr⁻¹, with the most rapid rate of C sequestration occurring in the first 25 to 30 year period (Qian and Follett, 2002).

The CENTURY model was used to determine the long-term effects of mowing and N management on soil organic C, because it is so costly and time consuming to collect soil C data (Bandaranayake et al., 2003; Qian et al., 2003). Clippings of Kentucky bluegrass collected under home lawn conditions were used to verify the model, and the model was used to predict biomass production and soil organic C and N. When clippings were returned for 10 to 50 years to the lawn, the model predicted that soil C sequestration would increase by 11 to 25% (Qian et al., 2003). When the golf course soils data (Qian and Follett, 2002) was used in the CENTURY model, results indicated that the turfgrass was serving as a C sink, with 23 to 32 Mg ha⁻¹ of soil organic C sequestered after 30 years (Bandaranayake et al., 2003). There are various estimates of C sequestration on grasslands established on previously disturbed soils, with the values between 0.33 Mg ha⁻¹ yr⁻¹ (Post and Kwon, 2002) and 0.9 Mg ha⁻¹ yr⁻¹ (Qian and Follett, 2003).

The use of mineral fertilizers and pesticides has contributed to C emissions, and has reduced the sustainability of urban lawns. A greater amount of C emission can be offset by reducing the use of management inputs from high mineral to zero or organic N inputs, reducing or eliminating the use of pesticides, reducing mowing, and returning clippings back to turfgrass soils. Due to the large area of land covered by turfgrass in North America, even a small change in management practices that reduce C emissions may be an important contribution to the global CO₂ level.

Tall fescue (*Festuca arundinacea* Schreb) (TF), and perennial ryegrass (*Lolium perenne* L) (PR) are used as cool season species in turfgrass lawns in the United States.

Both species are colonized by fungal endophyte (*Neotyphodium* spp) (Roberts et al., 2005). Endophytic fungi colonize leaf sheaths and stems, and are transmitted only through seeds. These fungi are non-toxic and do not cause any harm or disease in the grasses (Latch, 1997; Roberts et al., 2005). Endophytic fungi can support seed germination establishment, competitiveness, drought tolerance, summer survival, and insect and disease resistance to grasses (Latch, 1997). These benefits may lead to enhance biomass production and increase soil C sequestration (Franzluebbers et al., 1999). Salminen and Grewal (2002) and Salminen et al. (2003) have shown that practices, such as frequency mowing, have a significant impact on alkaloid production and accumulation in TF and PR.

Nitrogen and Turfgrass

Nitrogen is an essential element for turf establishment, reproduction, and chlorophyll, protein, and amino acid formation. The N requirement for individual plant species varies (Petrovic, 1990). Turfgrass yield and quality is largely controlled by rate of application of N via synthetic or organic fertilizers (Landschoot and Waddington, 1987; Robinsons and Birkenholtz, 2003; Carrow et al., 1987). Nitrogen fertilizers play an important role in maintaining shoot growth, color, quality, and grass health (Kerek et al., 2003). Plant uptake of N is a function of N fertilizer source, N release rate, and plant species; therefore plant N uptake efficiency varies from 5 to 74% (Petrovic, 1990). Mowing practices are a major source of N loss, via clipping removal. A study that evaluated clipping return showed that return of clippings increased dry matter yield from

30 to 72%, improved N uptake from 48 to 60%, and increased N use efficiency from 52 to 71% (Kopp and Guillard, 2002).

Fate of Nitrogen

Due to its role in maintaining turf quality, application of N fertilizer is one of the most common turfgrass management practices. Recently, environmental concerns over the fate of N fertilizer applications have stimulated research to quantify N fate under turfgrass systems. However, most research efforts have focused on maximizing efficiency of N applications by quantifying the N lost from the turfgrass system. Research on the fate of applied N has focused on traditional, widely used turfgrasses like Kentucky bluegrass (*Poa pratensis* L.) and to a lesser extent on creeping bentgrass (*Agrostis pa/ustris* Huds.), perennial ryegrass (*Lolium perenne* L.), and bermudagrass (*Cynodon dacty/on* L.) (Petrovic, 1990).

The fate of N applied to turfgrass depends on several factors including N release rate, N source, N rate, species, cultivar, clipping management, soil texture, and irrigation management (Petrovic, 1990). The five major components of the N cycle to determine the N fate in soils are mineralization (ammonification and nitrification), immobilization or assimilation (NO_3^- to organic N), volatilization (NH_3), denitrification (NO_2 , NO, and N_2), and N_2 fixation (Coyne and Frye, 2005). Nitrogen can accumulate in soils from N fixation via lightning or microorganisms (biological fixation), precipitation, plant decomposition, and organic or inorganic amendments.

Nitrogen uptake by turfgrass has primarily focused on N recovered in clippings and above ground cover. Fagerness et al. (2004) applied NH_4NO_3 to bermudagrass in a

controlled-environment growth chamber at a rate of 50 kg N ha⁻¹ three times during 12 weeks, and recovered 65% of the total N in plant tissue. Bowman et al., (2002) recovered 84% of applied N in hybrid bermudagrass. Miltner et al (1996) recovered 35% of applied N in Kentucky bluegrass clippings over a 2 yr period. Bowman et al. (1989) recovered 75% of applied N in Kentucky bluegrass at 5 d after treatment. Bristow et al (1987) applied ¹⁵N labeled NH₄NO₃ to perennial ryegrass and recovered, during four harvests, a total of 55% of applied N. Starr and De Roo (1981) recovered 35 and 20% of applied N in clippings after a May and September N application, respectively. Wesely et al. (1988) investigated N recovery in Kentucky bluegrass as N rate increased from 8 to 32 kg N ha⁻¹. There were no differences in percent of applied N recovered at N rates greater than 8 kg N ha⁻¹ (Wesely et al., 1988). Barraclough et al. (1985) applied NH₄NO₃ at high rates to perennial ryegrass and found that as N rate increased from 250 to 900 kg N ha⁻¹, percent of applied N recovered decreased dramatically to 50%.

Organic-N is mineralized to NH₃ then protonated chemically to form NH₄⁺. Ammonium is a stable form of N in the soil, and its loss as leachate is often negligible (Mazur and White, 1983). Mineralization of organic-N (the transformation of organic-N into mineral N such as NH₄⁺ or NO₃⁻) is influenced by several factors, including: N content of compost, C:N ratio, total root mass (Fornara et al., 2011), source of N (Finlay et al., 1992; Turnbull et al., 1995; Schimel and Chapin, 1996), soil pH (Yao et al., 2009), soil moisture content and soil temperature (Watts et al., 2007), soil texture (Hartl and Erhart, 2005), and CO₂ concentration (Castellanos and Pratt, 1981). Shi et al. (2006) measured N mineralization rates from bermudagrass clippings in a 28-d incubation study. They found that 20 to 30% of C and N respectively, in the clippings had been mineralized

by the end of the incubation study. Yao et al. (2009) measured total C and N mineralized in a 67-d laboratory incubation study contain four treatments as follows: (1) control, no mineral N input or addition of grass clippings, (2) N treatment, mineral N input at a rate of 60 mg NH_4NO_3 N g^{-1} soil, (3) clipping treatment, an addition of ≈ 1 cm-long grass clippings at 4 mg an oven-dry clippings g^{-1} soil, and (4) #2 and #3 treatments. They found that the addition of grass clippings significantly increased C and N mineralization by $\approx 58\%$ and 33% , respectively.

Ammonification is the first step of mineralization, and once it takes place, NH_3 is easily loss to the atmosphere through volatilization. Ammonia volatilization is more prevalent in soils with higher pH values (alkaline soils) and warmer climates (Clay et al., 1990; Ferguson and Kissel, 1986). Ammonia loss through volatilization has been shown to be highly variable and drastically reduced by watering immediately after the application of fertilizer in a fine- textured, saturated, and warm soil (Petrovic, 1990; Bowman et al., 1987). Irrigation or precipitation following N applications affects the position of N in the turfgrass and thereby influences volatilization. When N remains in the shoot and thatch region, volatilization potential is greater than if N is moved into the soil. Bowman et al. (1987) reported that 36% of applied N was volatilized when no irrigation followed the N application. Irrigation applications of 1 and 4 cm within 5 min of the N application reduced volatility losses to 8 and 1%, respectively. Sheard and Beauchamp (1985) found volatilization losses were reduced from 15 to 7% when a 1.2 cm rainfall occurred within 72 h of N application.

The presence or absence of a thatch layer influences volatility losses. The thatch layer has significant urease activity, which is necessary to convert urea to $\text{NH}_3\text{-N}$

(Bowman et al., 1987). Nelson et al. (1980) found volatilization losses of 39% of applied N from Kentucky bluegrass cores with a 5 cm thatch layer and only 5% of applied N from cores with no thatch layer. Wesely et al. (1987) determined volatilization losses of 35 and 31% from foliar applied N at a rate of 17 and 34 kg N ha⁻¹, respectively. The mechanism where foliar applied N is volatilized was elucidated by Torello et al. (1983). They indicated that high levels of urease activity in the thatch layer can cause rapid hydrolysis of applied urea, thereby increasing the pH of the water film on the thatch and turfgrass tissue and promoting volatilization.

The amount of NH₄⁺-N released from soil varies under different soil conditions. For example, Proctor et al (2010) reported that soils covered with fresh residue have stimulated NH₃ volatilization compared to non-covered soils. Burger and Venterea (2008) reported that less than 1% of applied N was lost by volatilization from loam and clay loam soils during an incubation study. Similarly, Knight et al. (2007) indicated that use of slow-release fertilizers had less than 2% N loss through volatilization over a 10-d period in a loamy sand and/or sand-peat (80:20) soils. Marshall et al. (2001) observed a 6% loss of applied N from volatilization and denitrification combined after a broiler litter application to tall fescue pastures at three different locations across southeastern U.S, including sites in the Coastal Plain (Alabama), Piedmont (Georgia), and Cumberland Plateau (Tennessee) physiographic regions.

Ammonium can be converted into NO₃⁻ during the nitrification processes. Nitrate-N is highly mobile in soils and may be lost via leaching, denitrification, and/or plant uptake (Johnson et al., 2006; Civeira and Lavado, 2007). Nitrate-N loss through leaching is influenced by soil texture (Brown et al., 1982; Bowman et al., 1998; Starr and DeRoo,

1981), irrigation rate, N-application rate, frequency and timing of fertilizer applications, rooting characteristics, and plant N requirements (Petrovic, 1990). Leaching losses of nitrate-N from golf courses have ranged from 1 to 56% of applied N (Sheard et al., 1985; Snyder et al., 1984), while leaching losses from lawns have been shown to be very small ($\leq 0.18\%$) (Miltner et al., 1996). Nitrate-N leaching may also be affected by season, and fall applications tend to have a lower rate of loss (Burger and Venterea, 2008). More than half of applied N may leach during the first 14 days after fertilizer application (Pare et al., 2008).

Nitrate-N loss may be controlled through management practices. Snyder et al. (1984) investigated the potential of tension-meter controlled irrigation systems to reduce N leaching on Pompano fine sand soils located at Lauderdale Research and Education center, Florida. Ammonium nitrate and sulfur-coated urea with a 30% 7-d dissolution rate were applied at a rate of $5 \text{ g N m}^{-2} \text{ mo}^{-1}$. Nitrogen loss ranged from 22 to 56% from the daily-irrigated NH_4NO_3 plots (no sensors) and from 2 to 8% from sensor irrigated plots. The study showed that sensor irrigation significantly reduced NO_3^- -N leaching from all N sources. Other studies have shown that NO_3^- -N leaching can be controlled through plant cultivar selection. A study investigated the effect of St. Augustine (*Stenotaphrum secundatum*) lawn versus a mixed species landscape and found that NO_3^- -N leaching losses were 4.1 and 48.3 kg N ha^{-1} , respectively (Erickson et al., 2001). In a greenhouse study, Bowman et al. (2002) found similar results when they compared six different turf species, indicating that St. Augustine was the most effective in preventing leaching losses. In a study conducted for three years, Guillard and Kopp (2004) investigated the potential of organic N sources on cool season grasses. Three treatments included

ammonium nitrate, polymer-coated sulfur-coated urea, and Sustane (slow release). Ammonium nitrate lost greater $\text{NO}_3\text{-N}$ through leaching compared to other treatments. Average leaching losses were 16.8, 1.7, and 0.6% of total applied N for NH_4NO_3 , polymer-coated sulfur-coated urea, and Sustane, respectively.

Nitrate-N can also be denitrified through microbial processes from NO_3^- to N gas, with subsequent loss to the atmosphere (Parsons et al., 1991; Mahimairaja et al., 1995). Denitrification rate may be affected by irrigation system (Christensen, 1983), rate of N inputs (Mahimairaja, 1995; Williams et al., 1992; Webster and Dowdell, 1982), addition of organic matter (Christensen, 1983; Mancino and Torello, 1986), soil moisture content (Mancino et al., 1988; Petrovic, 1990), and soil temperature (Mancino et al., 1988; Bijoor et al., 2008).

Nitrate-N may also be immobilized through microbial processes and become unavailable to plants in soils. Organic matter content, C:N ratio and soil microbial biomass all play important roles in $\text{NO}_3\text{-N}$ immobilization (Hadas et al., 1992). A C: N ratio of 19:1 or higher is required for immobilization to occur (Calderon et al., 2004).

Objectives

Studies addressing C sequestration, potential C and N mineralization, and decomposition rates in warm season turfgrass management systems in the southeastern U.S. are lacking. The objectives of the proposed study are to: 1) determine C sequestration under three major warm-season turfgrass species, 2) determine the effect of nitrogen fertilizer rate on CO_2 flux over time in hybrid bermudagrass, and, 3) determine decomposition rates and N release from turfgrass clippings.

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CHAPTER II

Carbon Sequestration under Warm Season Turfgrass in Home Lawns

Abstract

Turfgrass cover in the U.S. is expanding because of increasing urbanization and the addition of approximately 675,000 ha of residential property every year. Such perennial grass covers could have a significant effect on the soil carbon (C) cycle. Despite its large-scale presence in the urban ecosystem, the role of turfgrasses in C cycling in home lawns in southeastern US soils has not been documented, and studies in warm-season turfgrasses are lacking. The objective of this study was to estimate C sequestration in soil as affected by turfgrass species, including: bermudagrass (*Cynodon dactylon* (L.) Pers. x *C. transvaalensis* Burt Davy), centipedegrass (*Erecholmoa ophroides* (Munroe) Hack.), and zoysiagrass (*Zoysia japonica* Steud.). The study was initiated in the winter of 2012 and conducted for two years on a loamy sand (fine-loamy, kaolinitic, thermic Typic Kanhapludult) soil. Eighteen lawns were sampled twice per year, six lawns of each grass species, with the harvested grasses separated into stems/roots (rhizomes, stolons, roots), above ground biomass (verdure) + thatch, and belowground roots. Underlying soil samples (0-5, 5-10, and 10-20 cm) were also collected. Total C concentration was determined on finely ground oven-dried samples by combustion. Results showed an accumulation of organic C in the top 20 cm of the soil profile over the two-year period. In

this sampling period, C sequestration in soils was a minimum of 2.3 ± 0.2 (bermudagrass) to a maximum of 4.4 ± 0.5 (zoysiagrass) $\text{Mg ha}^{-1} \text{y}^{-1}$.

Introduction

Industrialization and urbanization have altered atmospheric composition, and increased greenhouse gas concentrations contribute to global climate change. Greenhouse gases include carbon dioxide (CO_2), water vapor (H_2O), ozone (O_3), methane (CH_4), and nitrous oxide (N_2O) (Ledley et al., 1999). Atmospheric CO_2 contributes about 25% to the greenhouse effect, while H_2O , O_3 , and CH_4 plus N_2O contribute $\approx 60\%$, $\approx 8\%$, and $\approx 7\%$, respectively (Karl and Trenberth, 2003).

Atmospheric CO_2 concentration has increased by $\approx 42\%$, from 280 ppmv in 1850 to 398 ppmv in 2014 (NOAA, 2014). This increase, and the probability of increasing global temperatures (NRC, 2011), is expected to alter distribution of carbon (C) between the atmosphere, vegetation and soils (Watson et al., 2000). Average global temperatures have increased by about 0.6 ± 0.2 °C since 1750, and 12 (2001-2012) out of the 14 warmest years on record have occurred since 1880 (NOAA-NCDC, 2012). Climatologists estimate that each 1000 Gt of C (≈ 3652 Gt CO_2) emissions increases global temperature on the average by ≈ 2 °C (NRC, 2011). Average global temperatures could increase by 2.5 ± 0.6 °C in the next fifty years (NRC, 2011). In addition to a sea level rise of 20 cm since 1870 (IPCC, 2007), there have been remarkable changes in the ocean's biology, permafrost (NRC, 2011), ecosystems (Greene and Pershing, 2007) and wildfire occurrence and extent (Running, 2006; NRC, 2011). These changes are caused by greenhouse gas emissions originating from human activities, including land use change,

soil cultivation, biomass burning and fossil fuel combustion (NRC, 2011). Schrag (2007) suggested that reducing temperature effects requires sequestration of C in the soil.

Soils are a major component of the global C cycle (Lal et al., 2004). Understanding the role of soils as a sink or a source for C on a global scale is critical for evaluating changes in atmospheric CO₂ concentration (Johnson and Curtis, 2001). Any disruption of soils due to land use or climate change can alter soil C pools and result in significant impacts on the C budget of the atmosphere. In contrast, increasing storage of C in the world's soils may reduce emissions of CO₂ (Rustad et al., 2000). Appropriate land use and soil management play major roles in reducing the greenhouse gases effect by increasing C stored within terrestrial ecosystems (Lal, 1997; Lal et al., 1998).

The conversion of forest to row crops results in depletion of SOC by an average of 35% (Post and Mann, 1990). Trumbore et al. (1995) found that when tropical dry forest in eastern Amazonia was converted to pasture it lost an estimated value of 130 kg SOC ha⁻¹yr⁻¹ from the top 10 cm of soil profile. In another part of eastern Amazonia, conversion of a tropical moist forest to pasture resulted in a loss of 300 kg SOC ha⁻¹ yr⁻¹ within the top 40 cm of soil (Desjardins et al., 1994). Similarly, Veldkamp (1994) observed that tropical wet forests of Costa Rica lost 900 kg SOC ha⁻¹ yr⁻¹ within the top 50 cm of soil, when converted to pasture. In contrast, Pregitzer and Palit (1996) observed that SOC increased by 600 kg ha⁻¹ yr⁻¹ within the top 70 cm of soil when an agricultural field was converted to oak forest in the Great Lakes region. Brown and Lugo (1990) reported that agricultural fields of Puerto Rico and the US Virgin Islands converted to a secondary forest, after 35 years, gained 800 and 1050 kg SOC ha⁻¹ yr⁻¹ within the top 25 and 50 cm of soil, respectively.

In addition, rates of soil organic carbon (SOC) sequestration under a subtropical wet forest plantation were higher than under a cool temperate-zone pine plantation, with an average accumulation of $338 \text{ kg C ha}^{-1} \text{ yr}^{-1}$ (Post and Kwon, 2000). However, perennial grasses are more effective than woody plants at sequestering C in soils (Post and Kwon, 2000). Baskin and Binkley (1998) reported that significant changes in C sequestration were found when sugarcane fields were converted to fast growing eucalyptus plantations. After 10 to 15 yr, SOC increased under eucalyptus in Hawaii by an average of 19.4 Mg ha^{-1} in the top 55 cm of soil. Thus, the amount of organic C stored in soils varies with ecosystems and land use change (Guo et al., 2006).

Carbon sequestration occurs through direct and indirect fixation of atmospheric CO_2 into various reservoirs such as geological, oceans, plant biomass, and soils. Soils are the second largest C reservoir (Schimel et al., 1994) with an estimated pool of 2500 Gt (Lal et al., 2004). The amount of C a particular soil can sequester varies and is partially dependent on climate factors, soil topography, and the ecosystem (Guo et al., 2006).

Plants fix atmospheric CO_2 via photosynthesis and turn it into biomass. This biomass enters the soil through decomposition of aboveground biomass, or belowground from roots and their exudates. This biomass is subsequently transferred to soil microbes and turned into soil organic carbon (SOC) through decomposition processes. Thus, roots play a significant role in SOC (Strand et al., 2008). Approximately 50% of the C fixed in photosynthesis is transported belowground and used for root growth, respiration, and/or assimilation to SOM (Nguyen, 2003). Roots participate in accumulation of SOC by their decomposition, and they supply C to soil by rhizo-deposition (Weintraub et al., 2007). Root exudation is the largest source of labile C inputs to the soil (Bertin et al., 2003).

Increased C content and turnover rates of roots may cause an increase of SOC accumulation following root decomposition (Matamala et al., 2003).

Most of the work on C sequestration has been conducted in row crops or forest eco-systems. Little research has focused on contributions from warm season turfgrasses such as bermudagrass, centipedegrass, and zoysiagrass. The turfgrass sector is a large industry with a sizable impact on the landscape of urban environments. The economic impact of turfgrass has been estimated to be \$62.2 billion in the US, in 2005 (Haydu et al., 2012). Nationally, the turfgrass industry generates 822,849 jobs, \$37.7 billion in labor income, and \$2.4 billion in indirect business taxes (Haydu et al., 2012).

Milesi et al., (2005) estimated the total U.S. turfgrass area to be $163,800 \pm 35,850$ km² (Table 1). Turfgrass cover in the U.S. is rapidly expanding because of increasing urbanization and the addition of approximately 700,000 ha of residential property each year (Robbins and Birkenholtz, 2003). These areas of extensive grass cover could have a significant effect on soil C sequestration. Soils developed under temperate grassland sequestered a C mass of 192 Mg ha⁻¹ compared to 127, 118, and 56.1 Mg ha⁻¹ for soils developed under cultivated agriculture, temperate forest, and desert scrub, respectively (Schlesinger, 1999).

Syntheses of historic soil testing data from golf courses in Denver and Fort Collins, CO, indicated that SOC sequestration in those soils occurred at a high rate of 1 Mg ha⁻¹ yr⁻¹ comparable to that of soil under the Conservation Reserve Program land (Qian and Follett, 2002). In addition to the soil, turfgrass thatch (the high organic matter layer which underlies the green shoots) is a temporary C sink (Raturi et al., 2005), and N in the thatch can affect C and N cycling.

A study of urban green space used two residential blocks in the Chicago area as the study site, with green space defined as ‘any soil surface capable of supporting vegetation’ (Jo and McPherson, 1995). Collected data included foliage, root and total biomass of numerous tree species, calculated using previously published biomass equations. Clippings were collected biweekly from 26 lawns, from Oct 1992 through September 1993, except in winter when mowing was not needed. Carbon storage in the grass was found to follow a predictable pattern, with maximum C in summer (July) followed by a decline into fall and winter (Jo and McPherson, 1995). Calculated turnover rates for stubble and root C were 2 and 2.9 years, respectively, indicating that about 50% of the stubble C and 34% of the total root C would be replaced each year. Average C concentration of the grass was 42.7% on a dry weight basis, with no difference in C concentration between live root, stubble or clippings. Over the year there was a net C release from turfgrass maintenance, with the greatest C release coming from mowing. Annual C released from mowing, irrigation and fertilization was 0.14 kg m^{-2} , while soil (0-60 cm) C storage was 20.6 kg m^{-2} , and organic C represented 79% of all stored C. Trees and shrubs represented 10.8 to 10.5% of that C, while the turfgrass presented 0.6%. Because of mowing, grass returned 1.5 times more C to the atmosphere than was sequestered. However, the studied green spaces were net sinks for C, owing to the soils and woody plants (Jo and McPherson, 1995).

A later study evaluated C sequestration under different turfgrass species (Qian et al., 2010). Cool season grasses such as fine fescue (both irrigated and non-irrigated), Kentucky bluegrass (irrigated), and creeping bentgrass (irrigated) were evaluated for changes in soil organic C, soil C sequestration and soil organic C decomposition. Four

years after establishment approximately 17 to 24% of soil organic C (0 - 10 cm depth) was contributed by the established turfgrass. The amount of soil organic C increase differed with turfgrass species and irrigation, with irrigated fine fescue having the highest soil organic C, which was estimated to be 3.4 Mg C ha⁻¹ year⁻¹. However, all turfgrass species sequestered C during the first four years after turf establishment (Qian et al., 2010).

There is limited research which examines C dynamics in the long-term, non-tilled setting that is warm-season turfgrass management. Warm season turfgrasses (when grown in the southeast United States) are additionally novel in that they are dormant after a severe frost, regaining color and growth in the spring. Also, many of our warm-season turfgrass species grow via above-and below-ground stem structures (stolons and rhizomes, respectively), and these structures affect carbohydrate storage within the plant, which may further affect C storage. Thus, there is a need to evaluate C dynamics in warm-season turfgrass lawns in the southeast United States. The objective of this study was to study C sequestration in warm-season turfgrasses managed in the home lawn environment, examining this sequestration as affected by grass species and time of year.

Methods and Materials

Study Sites

This two-year study (2012 and 2013) was conducted on selected home lawns located in Auburn, AL (32.598° N, 85.481° W). Every lawn was within 1.6 km radius from other locations, and all lawns were of a similar age (the neighborhood in which lawns were sampled largely had houses that dated to the 1950s and 1960s). From a survey of the homeowners, none of the lawns were new, and the averaged estimated age

of the sampled lawns was more than 20 years. In all cases the soil type was a Marvyn loamy sand soil (fine-loamy, kaolinitic, thermic Typic Kanhapludult). Three turfgrass species were included in the study: bermudagrass, centipedegrass and zoysiagrass. Cultivars were most likely: ‘common’ centipedegrass, ‘Tifway’ hybrid bermudagrass and ‘Emerald’ or ‘Meyer’ zoysiagrass, all typical and available cultivars for 20-year old home lawns. Six lawns of each turfgrass species were sampled, for a total 18 lawns. For this study, we had no control over mowing, clipping removal or fertilization procedures for these lawns. Observation and conversations with our participating homeowners revealed that most removed and discarded clippings with each mowing.

Turf and Soil Sampling Procedures

Each lawn was sampled twice a year, in the summer (July) and in the winter (January), when the lawns were dormant. At each sampling six samples (5.0 cm diam) were collected by hand from each lawn, with the samples taken randomly from each lawn to a final depth of 20 cm. Collected samples were taken fresh to the laboratory, where they were immediately separated into tissue and soil components as described next. Three of the samples were used for tissue sampling, with that sample separated into: 1) above ground growth (verdure) plus thatch, 2) stems (stolons and/or rhizomes), and, 3) roots. The remaining three samples were used for soil analysis, with those samples separated into the depth increments 0-5, 5-10, and 10-20 cm.

Soil samples were air dried and then ground to pass a 100-mesh sieve. Those samples were used for organic C analysis. Tissue samples were oven dried at 50°C for 48 hr and then ground (by hand) to pass a 100 mesh sieve for organic C analysis.

Total C concentration was determined on finely ground oven-dried stem (rhizomes, stolons, and roots), above-ground tissue of turfgrass plus thatch and soil samples via dry combustion using a LECO TruSpec CN (LECO Corp., St. Joseph, MI).

Bulk Density

Bulk density was determined from core samples collected with a slide hammer (AMS, Inc., Sampling Equipment, American Falls, ID), with adjustment for rock content using the USDA formula (Burt, 2004).

Carbon Sequestration calculation

Sequestered C was calculated as described by USEPA (2010) as follows:

$$C \text{ Mg ha}^{-1} = (\%C/100) \times BD \times D \times (10,000 \text{ m}^2 \text{ ha}^{-1})$$

Where:

% C = Mean percent of carbon content in soil

BD = Mean bulk density (in Mg m^{-3})

D = Soil depth (m)

Stored C in biomass calculated using C% and dry mass as follows:

$$C \text{ Mg ha}^{-1} = (\%C/100) \times \text{dry mass (Mg ha}^{-1}\text{)}.$$

Data Analysis

Differences in C sequestration due to grass species were determined via analysis of variance using the mixed model procedure (SAS Institute Inc., Cary, NC). Season of measurement and year were considered as repeated measures; depth of each lawn used as the main variable and fixed effect. Denominator degrees of freedom were calculated using the Kenward–Roger option. Means were compared using Pairwise Multiple Comparison Procedures (Tukey's Test) at $P < 0.05$.

Results and Discussion

Neither year nor season (winter or summer) had a significant effect on soil C in the year x grass x depth, year x depth, season x depth, or season x grass x depth interaction. This indicates that, within our relatively narrow two-year sampling period, soil C was relatively stable across year and month of sampling (Tables 2 and 3). The interaction of grass x depth was always significant, indicating that grass species sequestered C differently with depth (Tables 2 and 3). Regardless of sampling date and year, there were significant difference in soil C accumulation as affected by turfgrass species (Tables 2 and 3). Among all turfgrass species, C sequestered in underlying soil was the highest in zoysiagrass lawns and lowest in bermudagrass lawns. The amount of C stored in zoysiagrass lawns was estimated at $2.9 \text{ Mg ha}^{-1} \text{ yr}^{-1}$, four times higher than measured in bermudagrass lawns ($0.75 \text{ Mg ha}^{-1} \text{ yr}^{-1}$) in the top soil surface (0-5 cm) (Figure 1). Both bermudagrass and centipedegrass stored more soil C at the deepest depth (10-20 cm) than measured in the zoysiagrass (Figure 1). When stored soil C was summed for the three sampling depths zoysiagrass had the greatest stored soil C, followed by centipedegrass, and then bermudagrass (Figure 2). Lower soil C in bermudagrass may be partly due to more frequent mowing and clipping removal. Conversely, the greater shoot density of zoysiagrass may have created more biomass and a greater potential pool of C in that turfgrass.

Soil C sequestration in the entire sampled soil profile (0-20 cm) varied among grass species over the two-year sampling period (Table 4). Table 4 indicates significant variation in accumulated C among turfgrass species in the upper 20 cm of soil depth with both the year and season of sampling. For example, the amount of C stored (0-20 cm) in

zoysiagrass lawns at the January sampling was $2.4 \pm 0.3 \text{ Mg ha}^{-1} \text{ yr}^{-1}$, compared to 0.23 ± 0.03 and $2.2 \pm 0.24 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ in bermudagrass and centipedegrass lawns, respectively (Figure 2). The amount of stored C measured in July was 1.28 ± 0.14 , 2.13 ± 0.27 and $2.59 \pm 0.18 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ under bermudagrass, centipedegrass, and zoysiagrass, respectively, a greater amount than measured in winter (Figure 2). Carbon stored in soil under zoysiagrass was $4.4 \pm 0.5 \text{ Mg ha}^{-1} \text{ yr}^{-1}$, almost three times higher than that found in soil under bermudagrass, which was measured at $1.5 \pm 0.3 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ (Figure 2). This was comparable to the $1.4 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ stored in the top 20 cm of soil under ornamental lawns (Townsend-Small and Czimczik, 2010). The reason behind higher accumulation of organic C under zoysiagrass may be explained by combined effects of above-ground biomass (Figure 3) and below ground roots (Figure 4), which influence microbial activities and biomass deposition and/or decomposition in the top soil surface and may result in higher C sequestration in zoysiagrass compared to bermudagrass and centipedegrass.

Plant C is added to soil by deposition and decomposition of aboveground plant parts (litter decomposition) and belowground root exudates (Lemma et al., 2007). Carbon stored in roots was less than measured in the surrounding soil, and was greatest in the top layer (0-5 cm) (Figure 5). Major sources of SOC accumulation are belowground plant root activities (Tate et al., 1993) and aboveground biomass decomposition. Root densities of turfgrasses are expected to decline with depth, while decomposition of clippings and stems at or near the soil surface enriches the upper soil with C. These combined effects demonstrate why highest C accumulation in roots was measured in the 0-5 cm layer, with declines with depth (Figure 5). In all cases C sequestration in roots decreased with depth.

Greatest C storage was in zoysiagrass roots, followed by centipedegrass and then bermudagrass.

The intensity of root exudates varies with different compositions of plant tissues (Jackson et al., 2000). For example, more than 45% of the C assimilated by trees is ultimately transported belowground via root growth and root exudates, making soil as a significant sink of C (Montagnini and Nair, 2004). Reyes-Reyes et al., (2002) and Yelenik et al., (2004) observed an increase in the belowground C stock when grass dominated ecosystems were invaded by a tree population in the central highlands of Mexico. Similarly, silvo-pasture systems were observed to have a greater accumulation of C in soil when compared with adjacent pastures (tree free) in Florida (Haile et al., 2008).

Carbon sequestration in roots significantly ($P \leq 0.05$) varied by depth among grass species with season (Table 5) and year (Table 6). The two-way interactions grass x depth, grass x year, grass x depth and grass x year significantly affected C accumulation in the roots. The amount of C stored in grass roots in the 5 to 10 cm layer was higher in the zoysiagrass lawns ($99 \pm 9 \text{ kg ha}^{-1}$) compared to bermudagrass and centipedegrass lawns (18 ± 3 and $82 \pm 9 \text{ kg ha}^{-1}$, respectively) in January 2012 (Figure 5). The decline in C from roots continued as sampling depth increased. Differences in accumulated C content among grasses can be explained by the differences in grass root biomass and anthropogenic activities such as clipping frequency and fertilization, which impact the above ground biomass, decreasing the amount of C that will be transported belowground and partitioned in root growth.

Significant variation in accumulated C in roots among turfgrass species was found with both season and year (Table 7). The two-way interactions of grass x year and grass x season and the three-way interaction grass x year x season significantly ($P \leq 0.05$) affected C accumulation in grass roots. For example, C sequestered in zoysiagrass roots increased from 0.4 ± 0.02 (January 2012) to 0.7 ± 0.04 Mg ha⁻¹ (July 2103), a 75% increase in stored C. The amount of C accumulated (0-20 cm) in bermuadagrass roots increased from 0.09 ± 0.01 (January 2012) to 0.16 ± 0.01 (July 2103), an increase of 88% (Figure 6). Differences in accumulated C are directly related to sampling month and climate.

When actual root biomass was measured (Figure 7), both centipedegrass and zoysiagrass had greater rooting at all depths. The difference in rooting density at lower depths is probably the reason for the difference in C accumulation. The difference in SOC between the top and bottom soil layer was the highest (75%) in centipedegrass.

Significant variation in accumulated C among turfgrass species was found in stem tissue (Table 8). The two-way interaction grass x year and grass x year and the three-way interaction grass x year x season significantly ($P \leq 0.05$) affected C accumulation in stems. For example, C sequestered in stolons and rhizomes of zoysiagrass increased from 0.7 ± 0.03 (January 2012) to 0.9 ± 0.03 Mg ha⁻¹ (July 2103) a 22 % increase. The amount of C accumulated in stolon/rhizome of bermuadagrass increased from 0.33 ± 0.02 (January 2012) to 0.45 ± 0.03 (July 2103) a 27 % increase (Figure 8). Carbon stored in zoysiagrass rhizomes and stolons was 0.08 ± 0.01 Mg ha⁻¹ yr⁻¹, which is almost two times higher than in bermudagrass lawns, which was 0.05 ± 0.01 Mg ha⁻¹ yr⁻¹ (Figure 8). Centipedegrass sequestered less C in stem material than the other grasses (Figure 8). This is likely a

function of the reduced total weight of centipedegrass stems, because centipedegrass only has stolons, and not rhizomes. Differences in accumulation of C may be explained by the differences in stolon/rhizome biomass under all grasses (Figure 9).

Significant variation in accumulated C among selected turfgrasses in the aboveground biomass and thatch layer was found with season and year (Table 9). The two-way interaction grass x year significantly ($P \leq 0.05$) affected C accumulation in the aboveground biomass and thatch layer. For example, C sequestered in the aboveground biomass and thatch layer of zoysiagrass was the highest, $0.5 \pm 0.04 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ compared to 0.2 ± 0.02 and $0.2 \pm 0.03 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ in bermudagrass and centipedegrass, respectively (Figure 10). Carbon stored in the above ground biomass was 2.1 ± 0.2 , 1.8 ± 0.05 and $2.2 \pm 0.04 \text{ Mg ha}^{-1}$ (January of 2012), compared to 2.3 ± 0.27 , 2.0 ± 0.14 and 2.8 ± 0.13 (July 2013) Mg ha^{-1} under bermudagrass, centipedegrass, and zoysiagrass, respectively (Figure 10). From year to year C in thatch and verdure increased and average of 9, 10, and 18% under bermudagrass, centipedegrass, and zoysiagrass, respectively.

Statistical analyses indicated that turfgrass species, season and year sampling significantly ($P \leq 0.05$) affected C sequestration (Table 10). The two-way interaction grass x year affected significantly C accumulation in the top 20 cm of soil depth ($P = 0.04$). Zoysiagrass had highest levels of sequestered C with a value of 48.5 ± 3.2 , compared to 32.6 ± 3.6 and $42.2 \pm 4.2 \text{ Mg ha}^{-1}$ under bermudagrass and centipedegrass systems at the end of the study, respectively (Figure 11). The total amount of C accumulated varied from 2.3 ± 0.4 to $5.8 \pm 0.6 \text{ Mg ha}^{-1} \text{ yr}^{-1}$ under bermudagrass and zoysiagrass, respectively. Accumulated C content differences among grasses can be explained by the differences in grass biomass and human activities such as fertilization

and mowing frequency, in addition to environmental factors including soil and air temperature, soil moisture, and structure.

Conclusions

Grassland is a major component of the terrestrial ecosystem, comprising 27% of the total US area (Nickerson et al., 2011). These areas are vital to global C sequestration, and may act as sources or sinks for C at a significant level (Wohlfahrt et al., 2008). Carbon cycling in grassland ecosystems may be altered under land use change, species, climate change, and desertification (Conant and Paustian, 2002; Lal et al., 2004). Due to that large area of land covered by grass, even a very small change in management practices to reduce C emissions may be an important contribution to the global CO₂ level. For example, greater amount of C can be offset by reducing the use of mineral fertilizers to zero or organic N inputs, reducing or eliminating the use of pesticides, reducing mowing, and returning clippings back to the turfgrasses lawns. This study found that zoysiagrass had greater above and belowground biomass that resulted in greater C inputs to the soil than other warm season turfgrasses, such as centipedegrass and bermudagrass. Greatest SOC stock was found in the soil surface (0-5 cm) and the lowest was found in the greater depth (10-20 cm). Soil organic C contents of home lawns varied significantly with grass species, likely due to the individual or combined effects of species and plant density. However, more detailed research is needed on a number of aspects to make logical conclusions such as litter quality and quantity, C flux, and microbial dynamics with soil depth greater than 20 cm under home lawns.

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Table 1. Estimated turfgrass area by state (Milesi et al., 2005).

Turfgrass area (km ²)					
State	Area	SD†	State	Area	SD
Alabama	3130	611	Montana	735	149
Arizona	2559	619	Nebraska	1149	252
Arkansas	2098	421	Nevada	928	234
California	11159	2731	New Hampshire	1126	213
Colorado	2478	569	New Jersey	3942	943
Connecticut	2429	517	New Mexico	1545	315
Delaware	533	111	New York	6320	1450
District of Columbia	57	29	North Carolina	8112	1603
Florida	11570	2651	North Dakota	572	121
Georgia	5688	1160	Ohio	6733	1480
Idaho	942	191	Oklahoma	2689	605
Illinois	5729	1373	Oregon	1977	429
Indiana	3843	836	Pennsylvania	7293	1496
Iowa	2227	545	Rhode Island	506	116
Kansas	2004	449	South Carolina	4034	788
Kentucky	2446	489	South Dakota	692	137
Louisiana	3377	722	Tennessee	4201	863
Maine	975	182	Texas	13187	3055
Maryland	2471	542	Utah	1207	286
Massachusetts	4183	871	Vermont	524	97
Michigan	4538	1060	Virginia	4544	966
Minnesota	3176	690	Washington	3579	766
Mississippi	1969	393	West Virginia	1459	272
Missouri	3442	775	Wisconsin	3110	654
Total US				163,812	35867

†SE: standard error about the mean.

Table 2. Analysis of variance to compare C sequestration in soil as affected by sampling season and soil depth within grass species.

Effect	DF	F Value	Pr > F
Season	1	7.37	0.0079
Grass	2	6.43	0.0025
Season x Grass	2	0.11	0.8965
Depth	2	134.68	<0.0001
Grass x Depth	4	36.43	<0.0001
Season x Depth	2	1.18	0.3114
Season x Grass x Depth	4	0.22	0.9268

Table 3. Analysis of variance to compare C sequestration in soil as affected by sampling year and soil depth within grass species.

Effect	DF	F Value	Pr > F
Year	1	53.29	<0.0001
Grass	2	18.47	<0.0001
Year x Grass	2	3.50	0.0344
Depth	2	213.11	<0.0001
Grass x Depth	4	57.64	<0.0001
Year x Depth	2	2.16	0.1212
Year x Grass x Depth	4	1.23	0.3018

Table 4. Analysis of variance to compare C sequestration as affected by sampling season and year (0 to 20 cm depth) within grass species.

Effect	DF	F Value	Pr > F
Year	1	46.82	<0.0001
Grass	2	152.90	<0.0001
Grass x Year	2	3.07	0.0648
Season	1	10.25	0.0038
Grass x Season	2	0.15	0.8597
Grass x Year x Season	3	0.20	0.8939

Table 5. Analysis of variance to compare C sequestration in roots as affected by sampling season and soil depth within grass species.

Effect	DF	F Value	Pr > F
Season	1	11.54	0.0010
Grass	2	12.06	<0.0001
Season x Grass	2	1.84	0.1650
Depth	2	226.33	<0.0001
Grass x Depth	4	20.05	<0.0001
Season x Depth	2	2.37	0.0994
Season x Grass x Depth	4	0.57	0.6818

Table 6. Analysis of variance to compare C sequestration in roots as affected by sampling year and soil depth within grass species.

Effect	DF	F Value	Pr > F
Year	1	146.40	<0.0001
Grass	2	56.62	<0.0001
Year x Grass	2	30.72	<0.0001
Depth	2	641.00	<0.0001
Grass x Depth	4	56.78	<0.0001
Year x Depth	2	8.23	0.0005
Year x Grass x Depth	4	0.90	0.4680

Table 7. Analysis of variance to compare C sequestration in roots as affected by sampling season and year (0 to 20 cm depth) within grass species.

Effect	DF	F Value	Pr > F
Season	1	177.42	<0.0001
Grass	2	2781.55	<0.0001
Season x Grass	2	28.27	<0.0001
Year	1	795.01	<0.0001
Grass x Year	2	166.81	<0.0001
Season x Grass x Year	3	24.45	<0.0001

Table 8. Analysis of variance to compare C sequestration in stems (stolon/rhizome) as affected by sampling season and year within grass species.

Effect	DF	F Value	Pr > F
Year	1	295.27	<0.0001
Grass	2	4101.00	<0.0001
Year x Grass	2	12.80	<0.0001
Season	1	276.86	<0.0001
Grass x Season	2	57.63	<0.0001
Grass x Year x Season	3	9.77	<0.0001

Table 9. Analysis of variance to compare C sequestration in aboveground biomass and thatch as affected by sampling season and year within grass species.

Effect	DF	F Value	Pr > F
Year	1	45.01	<0.0001
Grass	2	64.66	<0.0001
Grass x Year	2	5.80	0.0066
Season	1	12.73	0.0010
Grass x Season	2	1.47	0.2442
Grass x Year x Season	3	0.25	0.8636

Table 10. Analysis of variance to compare total C sequestration (soil+tissue) as affected by sampling season and year within grass species.

Effect	DF	F Value	Pr > F
Season	1	12.83	0.0015
Grass	2	173.95	<0.0001
Grass x Season	2	0.20	0.8234
Year	1	56.39	<0.0001
Grass x Year	2	3.58	0.0436
Grass x Year x Season	3	0.23	0.8774

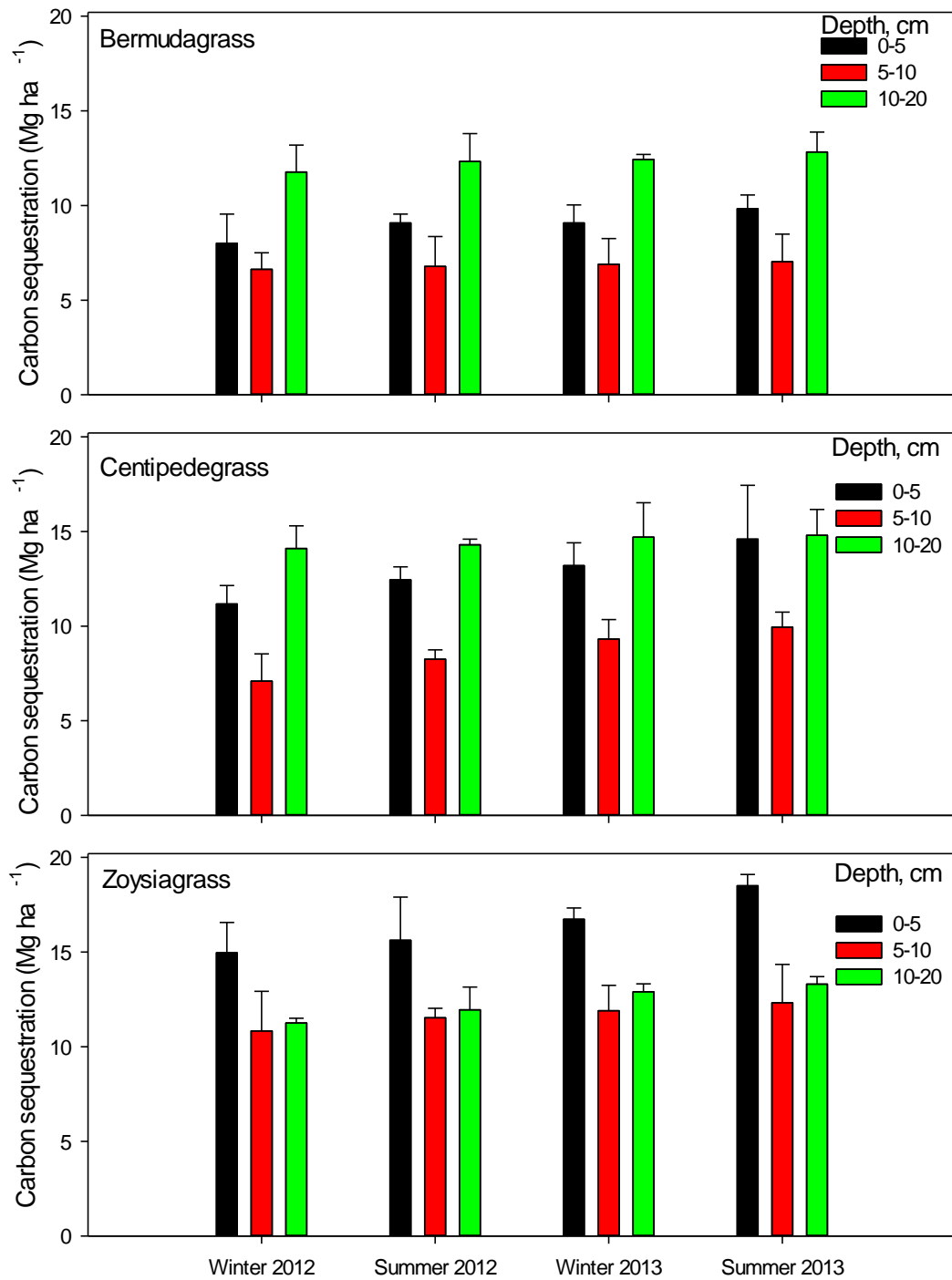


Figure 1. Carbon sequestration (Mg ha⁻¹) in soil as affected by turfgrass species, soil depth, and season of sampling. Bars represent standard errors about the mean.

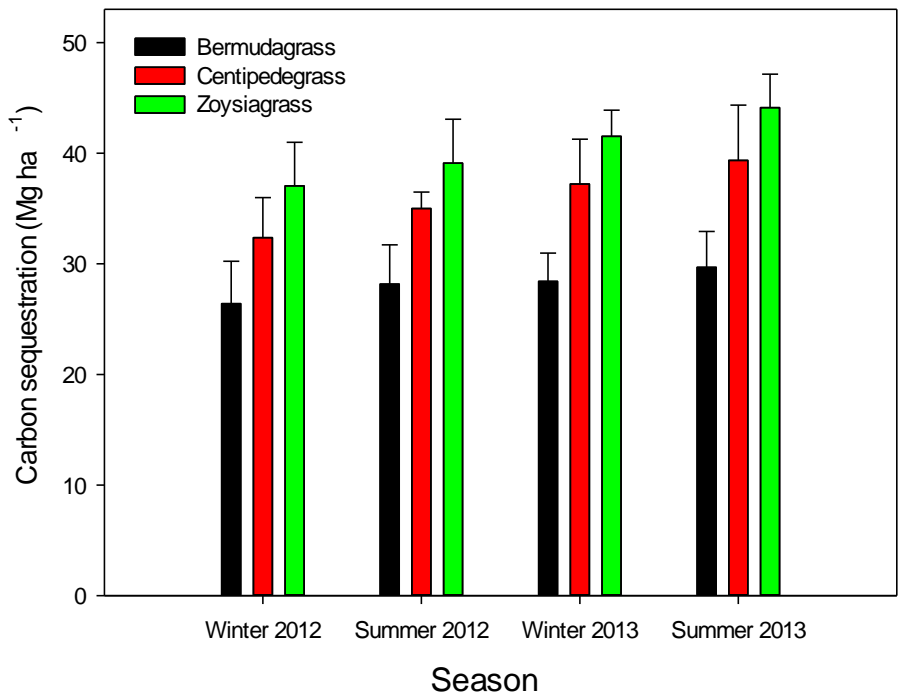


Figure 2. Carbon sequestration (Mg ha⁻¹) in the 0-20 cm soil layer as affected by turfgrass species and season of sampling. Bars represent standard errors about the mean.

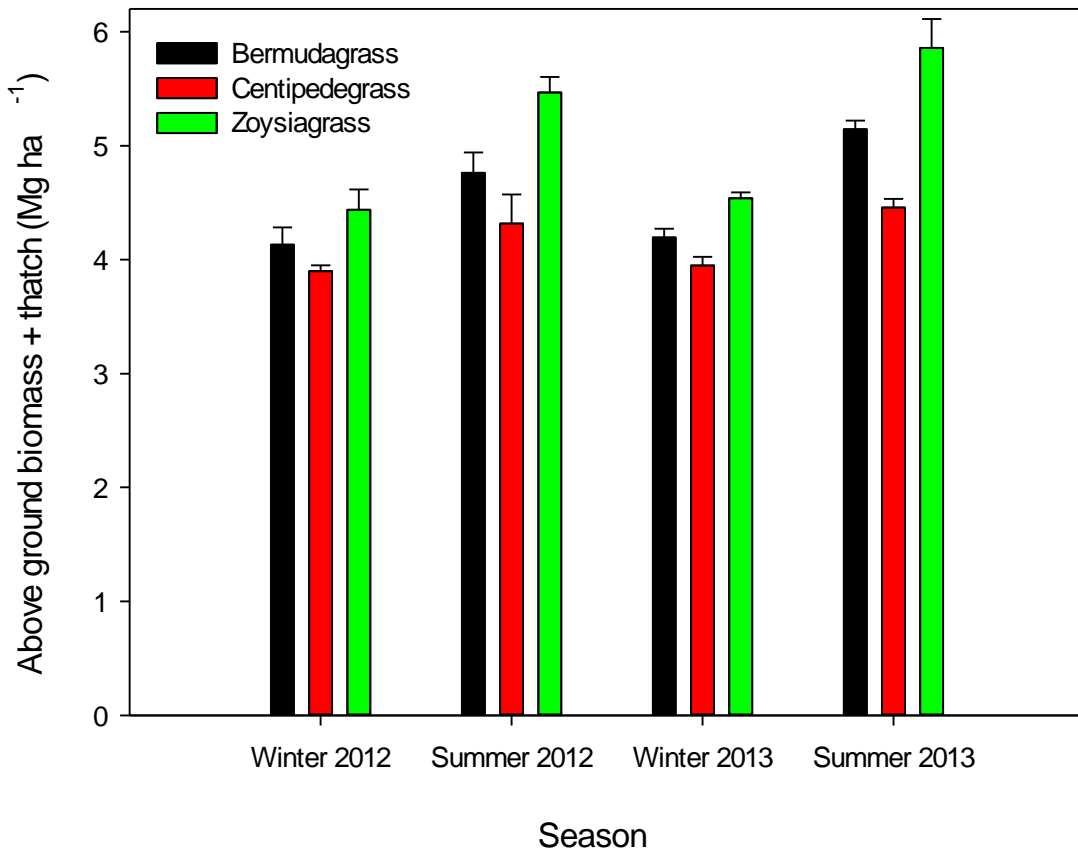


Figure 3. Above ground biomass (verdure) plus thatch (Mg ha⁻¹) harvested from sampled turfgrasses. Bars represent standard errors about the mean.

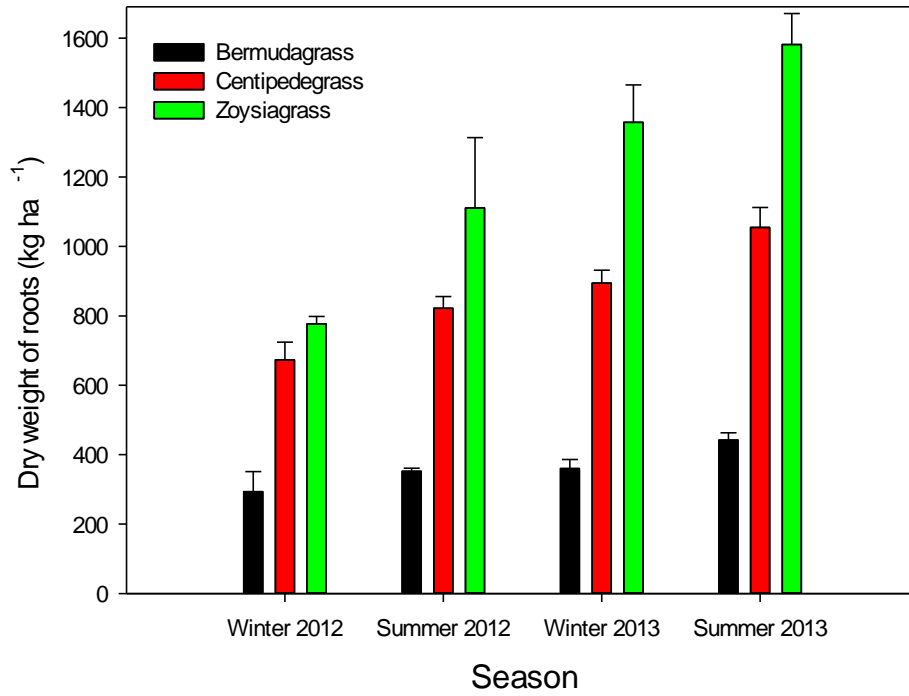


Figure 4. Root biomass (kg ha⁻¹) yield under selected turfgrasses in the 0 to 20 cm depth. Bars represent standard errors about the mean.

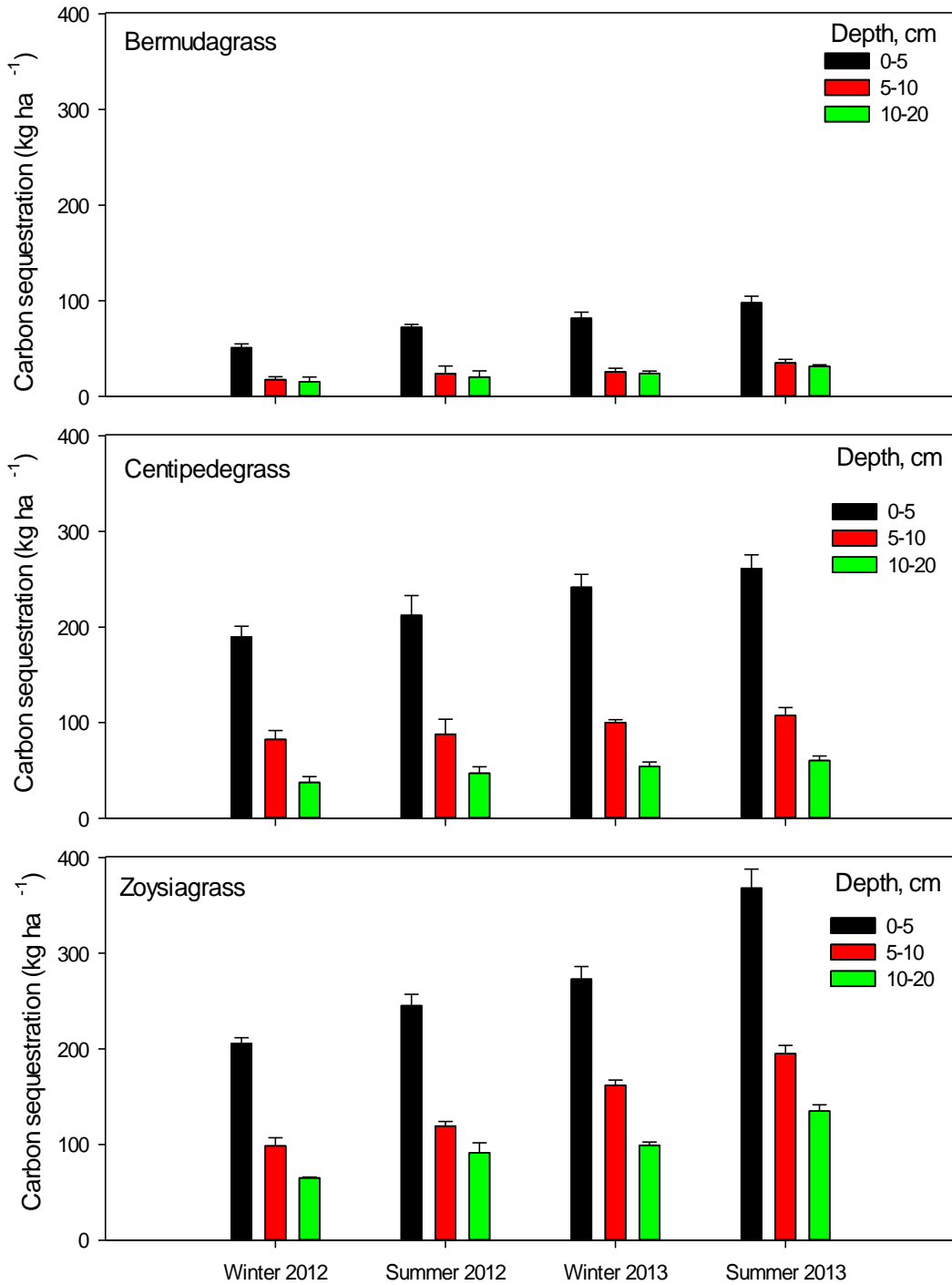


Figure 5. Effect of grass species, soil depth, and season of sampling on carbon concentration (kg ha⁻¹) in grass roots. Bars represent standard errors about the mean.

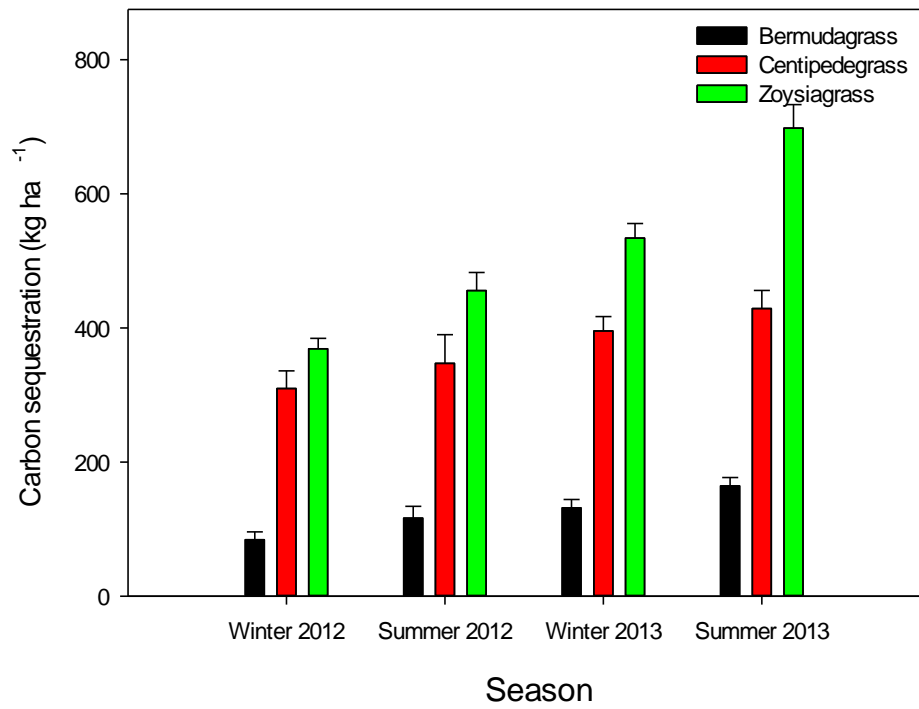


Figure 6. Carbon sequestration (kg ha⁻¹) in turfgrass roots in the 0 to 20 cm depth. Bars represent standard errors about the mean.

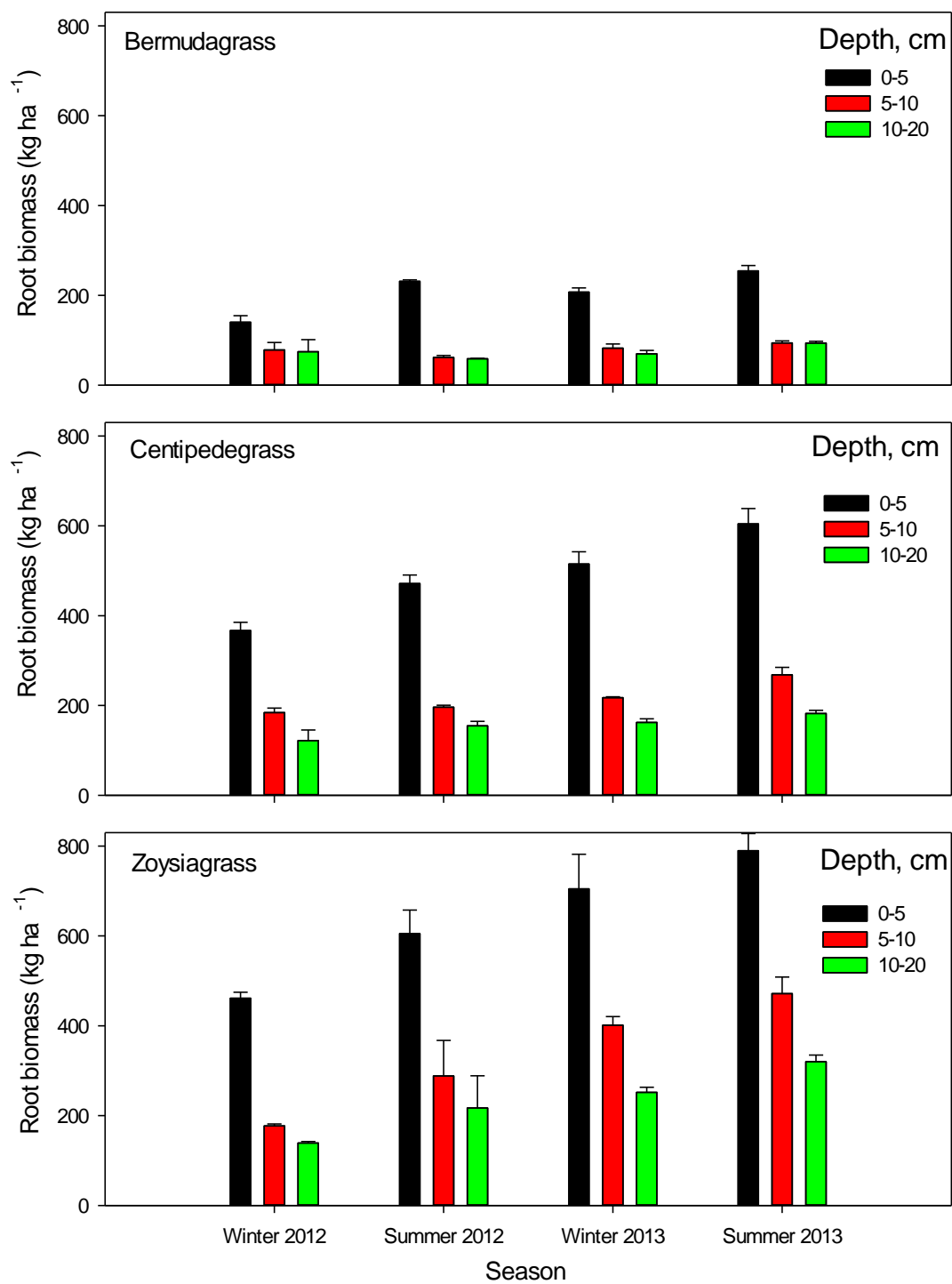


Figure 7. Effect of grass species, soil depth, and sampling season on root biomass yields (kg ha⁻¹). Bars represent standard errors about the mean.

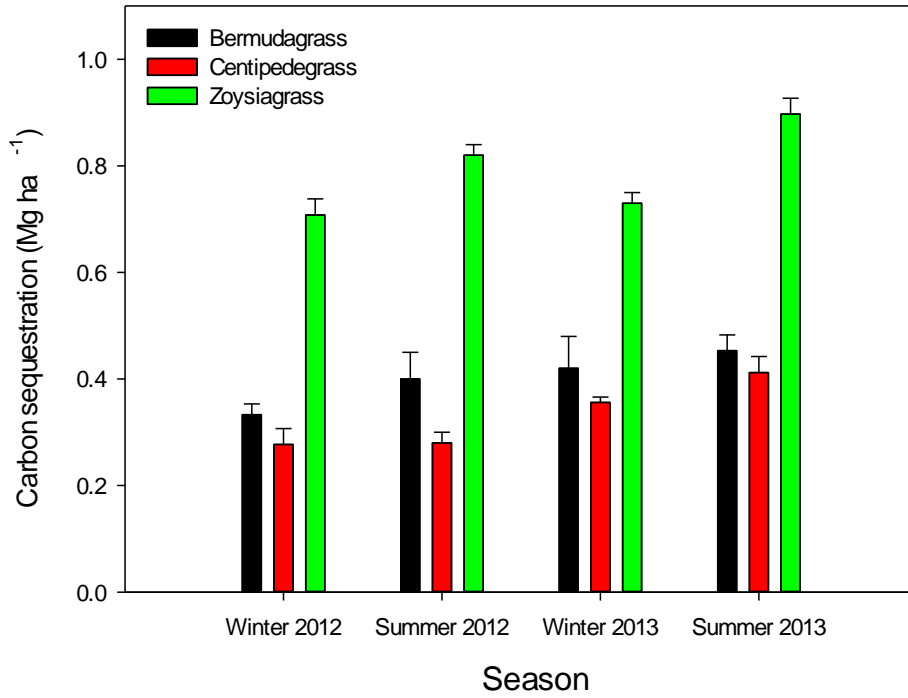


Figure 8. Effect of grass species and sampling season on carbon sequestration (Mg ha^{-1}) in turfgrass stems (rhizome and/or stolon). Bars represent standard error about the mean.

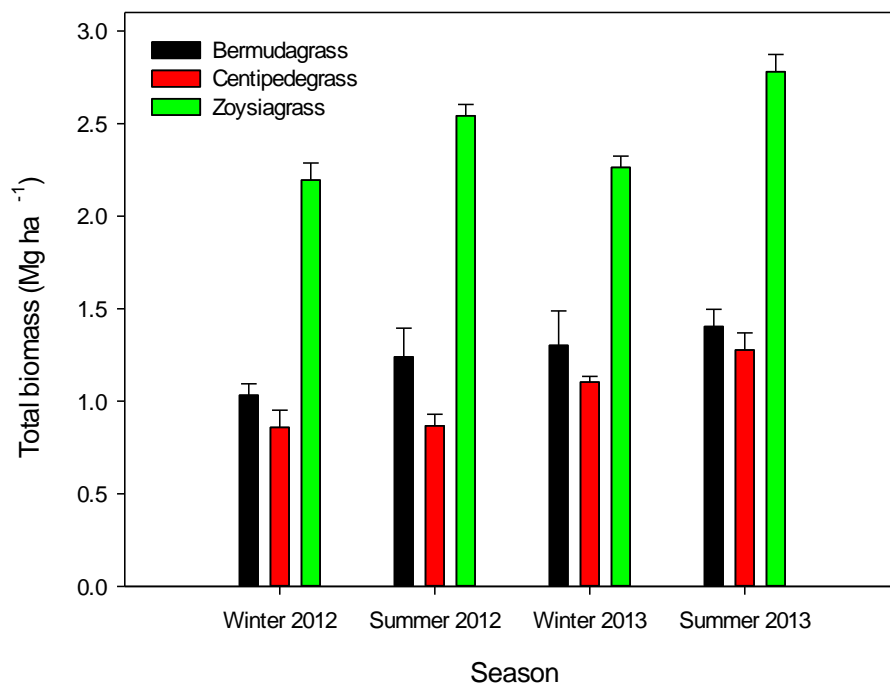


Figure 9. Rhizome/stolon biomass (Mg ha⁻¹) under sampled turfgrasses at different seasons. Bars represent standard error about the mean.

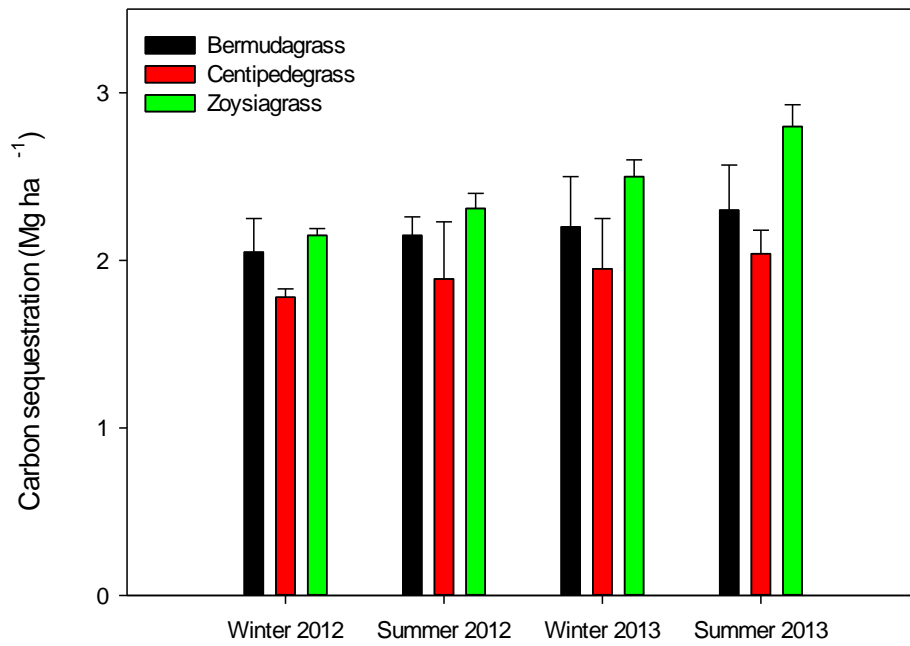


Figure 10. Effect of grass species and sampling season on carbon sequestration (Mg ha⁻¹) in aboveground biomass and thatch layer. Bars represent standard errors about the mean.

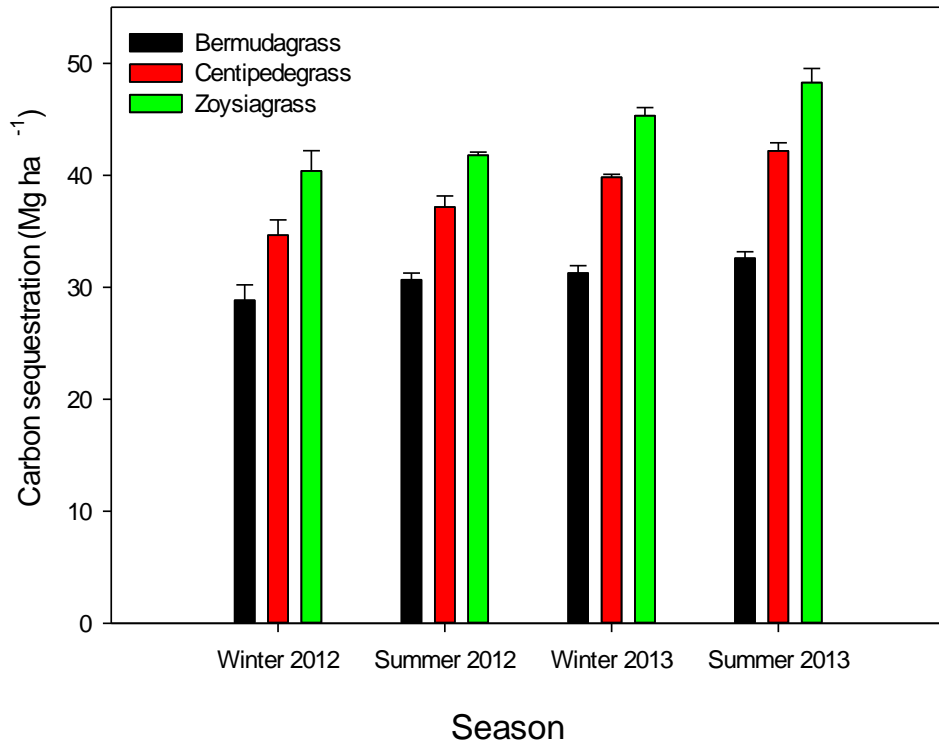


Figure 11. Total carbon sequestration (Mg ha⁻¹) (tissue plus soil) in 0 to 20 cm sampling depth. Bars represent standard error about the mean.

CHAPTER III

Carbon Dioxide Flux from Bermudagrass Turf as Affected by Nitrogen Rate

Abstract

Increasing greenhouse gas concentrations have contributed to global climate change. Greenhouse gases include carbon dioxide (CO₂) and other gases. Atmospheric CO₂ concentration has increased by $\approx 42\%$, from 280 ppmv in 1850 to 398 ppmv in 2014. This increase, and the probability of increasing global temperatures, is expected to alter the distribution of carbon (C) between the atmosphere, vegetation and soils. Despite its large-scale presence in the urban ecosystem, the role of turfgrasses in C cycling has received limited attention, and studies with warm-season turfgrasses are especially lacking. The objective of this study was to estimate CO₂ flux from soil as affected by nitrogen (N) applied to bermudagrass (*Cynodon dactylon* (L.) Pers. *C. transvaalensis* Burt Davy). The study was initiated in March, 2012 on eight-year-old 'Tifway' hybrid bermudagrass plots located at the Auburn University Turfgrass Research Unit (32.58° N, 85.50° W) on a Marvyn loamy sand (fine-loamy, kaolinitic, thermic Typic Kanhapludult) soil. The experimental design was a randomized complete block with four N rates of 24, 49, 98, and 196 kg N ha⁻¹ yr⁻¹ that were replicated three times. Carbon dioxide flux was measured weekly for 95 weeks using an automated soil CO₂ flux system (LiCor LI-8100A). Soil temperature and moisture were also determined during CO₂ flux

measurements. Results showed strong correlation between CO₂ flux and N rate ($r^2 = 0.99^{**}$). Carbon dioxide flux during the study period increased from 107 ± 4.5 to $144 \pm 4.4 \text{ Mg ha}^{-1}$ as N rate increased from 24 to 196 kg N ha^{-1} , respectively.

Introduction

Increasing atmospheric carbon dioxide (CO₂) concentration is contributing to global climate change. Atmospheric CO₂ contributes about 25% to the greenhouse effect, while H₂O, O₃, and CH₄ plus N₂O contribute $\approx 60\%$, $\approx 8\%$, and $\approx 7\%$, respectively (Karl and Trenberth, 2003).

Atmospheric CO₂ concentration has increased by $\approx 42\%$ from 280 ppmv in 1850 to 398 ppmv in 2014 (NOAA, 2014). This increase, and the probability of increasing global temperatures (Rustad et al., 2000), is expected to alter distribution of carbon (C) between the atmosphere, vegetation and soils (Watson et al., 2000). Average global temperatures have increased by about 0.6 ± 0.2 °C since 1750, and 12 (2001-2012) out of the 14 warmest years on record have occurred since 1880 (NOAA-NCDC, 2012). Increasing global temperatures are positively correlated to rising sea levels. Melting icecaps and glaciers are estimated to raise average sea levels by 15 to 100 cm by the end of the 21st century (Douglas, 2004). Increasing sea levels disrupt the marine ecosystem as a result of changing ice cover, salinity, oxygen levels, and water temperature (IPCC, 2007). In addition to a sea level rise of 20 cm since 1870 (IPCC, 2007), there have been remarkable changes in the ocean's biology, permafrost (NRC, 2011), ecosystems (Greene and Pershing, 2007) and wildfire occurrence and extent (Running, 2006; NRC, 2011). These changes are caused by greenhouse gas emissions originating from human

activities, including land use change, soil cultivation, biomass burning, and fossil fuel combustion (NRC, 2011). Climatologists expect that average global temperatures could increase by 2.5 ± 0.6 °C in the next fifty years (NRC, 2011).

Soils are a major component of the global C cycle (Lal et al., 2004). Appropriate land use and soil management play major roles in reducing the greenhouse effect by increasing C stored within terrestrial ecosystems (Lal, 1997). Management practices such as cropping intensity, rotation, and tillage practices can increase or decrease CO₂ emission, depending on the management activity. Reicosky and Lindstrom (1993) and Reicosky et al. (1999) suggested that some intensive tillage disrupted soil aggregation and incorporated plant residue, which could decrease SOC. However, perennial grasses are more effective than woody plants at sequestering C in soils (Post and Kwon, 2000).

Soil respiration is a composite flux that includes respiration of soil organisms, organic matter decomposition, and plant roots (Doff et al., 2004; Subke et al., 2006). Root respiration contributes more CO₂ to total soil respiration than all other processes (Hanson et al., 2000) with estimated values between 30 and 70% in temperate forests (Buchmann, 2000), 50 and 93% in tundra ecosystems (Raich and Tufekcioglu, 2000), and 17 and 60% in grasslands (Kucera and Kirkham, 1971; Coleman, 1973; Herman, 1977; Buyanovsky et al., 1987; Dugas et al., 1999). In fact, there are many factors controlling CO₂ flux rates, including but not limited to: 1) temperature, 2) soil moisture, 3) vegetation and substrate quality, 4) plant rooting density, 5) soil physical and chemical properties, and, 6) land-use change. Of these factors temperature and soil moisture are considered the most influential environmental factors affecting soil CO₂ flux rates due to their influence on terrestrial ecosystem productivity and the decomposition rate of SOC

(Schlesinger 1977, Raich and Schlesinger, 1992). Soil respiration is often inhibited when soil temperatures and water content are low (Wildung et al., 1975). In contrast, respiration in saturated soils is due to anaerobic microbial activity, which is less effective at decomposing organic matter (Fisher and Binkley, 2000). Edwards (1975) studied changes in temperature and moisture contents on CO₂ flux rates from forest litter and a mixed deciduous forest floor in Tennessee. A strong relationship was found between mean daily respiration rates and daily temperature change. Overall, moisture content had the greatest effect on soil CO₂ rates, both during heavy rains or drought periods (Edwards, 1975). Thus, soil moisture content influences soil respiration, and dry soils typically respire less than their wet counterparts (Davidson et al., 2000; Pangle and Seiler, 2002; Reichstein et al., 2002; Jin et al., 2013). Low soil moisture suppresses soil microbial populations (Schimel et al. 1999; Moyano et al., 2012) and their activities (Grahammer et al., 1991), limiting soil respiration. Rochette et al. (1990) found that soil respiration increased by 90% following a rainfall preceded by a long dry period. Furthermore, Jin et al., (2013) found that soil respiration in an Alfisol increased by 50% when soil water holding capacity increased from 25% to 50%.

Temperature is the most important environmental factor affecting soil respiration (Kirschbaum, 1995; Davidson et al., 1998), and soil respiration increases with increasing temperature (Raich and Schlesinger, 1992; Rustad and Fernandez, 1998). Soil warming had a greater effect on CO₂ emitted from soils in the boreal and tundra region than from that emitted from temperate and tropical regions (Niklinska et al., 1999) due to a greater concentration of labile soil C in the boreal and tundra soils (Schlesinger and Andrews, 2000). Maier and Kress (2000) reported a strong correlation ($r^2 = 0.70$) between soil CO₂

flux rates and soil temperature in the top 7 cm of soil depth. Soil respiration in soils developed under tropical lowland forests was estimated to be $1092 \text{ g m}^{-2} \text{ yr}^{-1}$ compared to 662, 544, and $200 \text{ g m}^{-2} \text{ yr}^{-1}$ for soils developed under temperate forests, cultivated land, and desert scrub, respectively (Raich and Schlesinger, 1992). Globally, CO_2 flux is estimated to range from 68×10^{15} to $77 \times 10^{15} \text{ g C yr}^{-1}$ (Schlesinger and Andrews, 2000).

Fertilizer is also a major factor that influences belowground processes which affect soil respiration. Kowalenko et al., (1978) measured CO_2 flux under 2 soil types located in fallow fields that were fertilized at rates of 0, 90, 225, and 400 kg N ha^{-1} (ammonium nitrate) for 3 yr. They observed daily average CO_2 flux rates were lower in fertilized plots compared to control plots during measurement. The effects of urea fertilizer on CO_2 flux in a 26 yr-old Florida slash pine (*Pinus elliottii* var. *elliottii* Englem.) plantation was investigated by Castro et al., (1994) using static gas chambers. The study included fertilized and non-fertilized plots, where fertilized plots received annual application of urea at a rate of 180 kg N ha^{-1} for 5 yr before CO_2 flux measurements were performed. After 5 yr CO_2 flux rates in fertilized plots did not differ from non-fertilized plots (Castro et al., 1994). The response of soil CO_2 efflux rates in a 31-year old red pine (*Pinus resinosa* Ait.) plantation in northern Wisconsin to fertilization was examined by Haynes and Gower (1995) in a loamy fine sand (Sandy, mixed, frigid, entic Haplothod) soil. Fertilizer was applied at 150 kg N ha^{-1} twice a year, for three years, and CO_2 flux was measured monthly. During the 3 yr of measurement CO_2 flux rates were significantly lower in fertilized plots than in control plots.

In addition to soil temperature, moisture content, and fertilization, soil respiration may be affected by other factors such as root biomass and microbial population

(Buchmann 2000; Li et al., 2004; Dilustro et al., 2005). Li et al. (2004) measured CO₂ efflux and microbial biomass at soil depth of 25 cm in a secondary forest dominated by *Tabebuia heterophylla* and in a pine (*Pinus caribaea*) plantation in the Luquillo Experimental Forest of Puerto Rico. In that study four treatments were included: 1) root exclusion, 2) litter exclusion, 3) mixed root and litter exclusion, and, 4) control plots without litter or root exclusion. Soil CO₂ efflux decreased significantly in the control plots in both the pine plantation and secondary forest. When litter was excluded the effect on soil CO₂ efflux was greater in the pine plantation than in the secondary forest. Soil CO₂ efflux rates decreased from 2.33 ± 0.12 to 0.75 ± 0.06 g C m⁻² d⁻¹ and from 2.66 ± 0.16 to 1.21 ± 0.11 g C m⁻² d⁻¹ under litter exclusion treatment in the plantation and the secondary forest, respectively. In comparison, total soil microbial biomass was lowest under the mixed litter and root exclusion plots. The exclusion of roots decreased total microbial biomass from 711 ± 43 to 489 ± 22 mg C kg⁻¹ soil and from 618 ± 45 to 256 ± 28 mg kg⁻¹ soil in the plantation and the secondary forest, respectively (Li et al., 2004).

Although CO₂ has been widely studied in agricultural and forestry settings, the phenomena has been less studied in turfgrass systems. The perennial nature and high C content of turfgrass systems are unlike many agricultural crops, especially as turfgrass is managed intensively as a long-term crop with little cultivation. Thus, the objective of this study was to examine CO₂ flux from an established sward of hybrid bermudagrass which had received 5 yr of N fertilization, at varying rates.

Materials and Methods

Site Description

The study was initiated in March, 2012 on eight-year-old ‘Tifway’ hybrid bermudagrass plots located at the Auburn University Turfgrass Research Unit (32.58° N, 85.50° W) on a Marvyn loamy sand (fine-loamy, kaolinitic, thermic Typic Kanhapludult) soil. Soil pH was 5.8, and background yearly soil tests were used to apply P and K according to soil-test recommendations of Auburn University Soil Testing Laboratory. The research plot area was managed as a golf course fairway with mowing 3 of 7 days at 2.5 cm mowing height. Supplemental irrigation was provided in the absence of rainfall so that irrigation plus precipitation equaled 2.5 cm wk⁻¹. Plots were mowed with a walk behind reel mower and clippings were not removed. The experimental design was a randomized complete block with four N rates of 24, 49, 98, and 196 kg N ha⁻¹ yr⁻¹ that were replicated three times. Nitrogen was applied as urea (46-0-0) in split applications five times monthly in April 27 (day 118), June 1 (day 153), July 11 (day 191), August 10 (day 223), and September 6 (day 250) of 2012 and in May 6 (day 126), June 10 (day 161), July 15 (day 196), August 12 (day 224), and September 13 (day 256) of 2013. These N rates had also been applied for 5 years prior to the initiation of this experiment. Plot size was 1.5 x 3 m. Initial soil organic C contents (g kg⁻¹) to a depth of 10 cm were 20.8±1.8 (24 kg N ha⁻¹), 23.1 ± 1.3 (49 kg N ha⁻¹), 26.7 ± 2 (96 kg N ha⁻¹), and 28.5 ± 1.7 (196 kg N ha⁻¹).

Carbon Dioxide Flux Sampling

Carbon dioxide flux was measured weekly for 95 weeks from March 7, 2012 to December 25, 2013 using an automated soil CO₂ flux system-Infra Red Gas Analyzer (IRGA) (LI-COR, LI-8100A, LICOR, Inc., Lincoln, NE). A long term (8100-104) soil gas chamber was attached to the LI-COR system and placed on a 20 cm diam. PVC

collars that were inserted 3 cm deep in each plot. Each plot had 2 collars randomly placed for a total of 24 collars. Carbon dioxide fluxes were measured between 12:00 and 2:00 p.m. on sampling days. Using an auxiliary sensor, soil temperature and moisture were simultaneously determined during the period that CO₂ flux measurements were collected.

Cumulative CO₂ flux was calculated by linear interpolation based on the CO₂ flux at each sampling time. Significant effects were identified by analyses of variance as implemented in SAS 9.2 using PROC MIXED procedures and maintaining blocks as a random effect (SAS, 2007). Effects were considered significant at $P < 0.05$. Regression analysis, means and standard errors were obtained using SigmaPlot 12.3 Statistical Software (Aspire Software International, Ashburn, VA, USA).

Results and Discussion

Environmental Conditions

During the experimental period average air temperatures for 2012 were slightly warmer (21°C) than that observed in 2013 (19°C), due to warmer temperatures from June to August, 2012 (average of 30.5°C). Air temperatures from January to April were similar in 2012 and 2013: 15.9 and 13.7°C, respectively. Average annual soil temperature was 20.1 ± 1.4 °C over the two-year period, with a mean average soil temperature from December to February of 11.5 ± 1.1 °C, from March to May of 19.1 ± 2.4 °C, from June to August of 28.1 ± 3.0 °C, and from September to November of 20.8 ± 1.1 °C.

Precipitation had a regular rainfall pattern during the two-year period, with few periods of dry weather. Dry periods occurred in October and November whereas

February, April, June, July, and December were wet. Over the study the total number of wet days (>1mm rainfall) was estimated to be 71 and 103, with an annual rainfall amount of 1163 and 1536 mm in 2012 and 2013, respectively. The summer (510 mm of rain) and winter (667 mm) seasons in 2013 were wetter than the summer (288 mm) and winter (426 mm precipitation) seasons of 2012. Lower precipitation was observed in fall of 2012 and 2013 with estimated cumulative values of 163 and 99 mm, respectively. However, accumulated precipitation in 2013 (1523 mm) was greater than that in 2012 (1163 mm) by 24%. Although precipitation data was a factor in the experiment it should be noted that irrigation was supplemented as needed in any dry periods so that precipitation plus irrigation totaled 2.5 cm wk⁻¹.

General Trends of CO₂ Flux

In general, statistical analyses indicated that increasing N rate from 24 to 196 kg ha⁻¹ significantly ($P < 0.05$) increased CO₂ flux from 323 to 426 kg ha⁻¹ d⁻¹ during 2012 and from 254 to 354 kg ha⁻¹ d⁻¹ during 2013, respectively (Tables 11 and 12). Daily fluxes of CO₂ ranged from 59 kg ha⁻¹ day⁻¹ in the winter of 2012 (December) to 893 kg ha⁻¹ day⁻¹ in the summer of 2013 (June) under 24 and 196 kg N ha⁻¹, respectively (Figures 12 - 15). Fluxes of CO₂ increased among treatments after N application, but returned to typical values within 2 weeks. For example, during 2012, at one week after N application (d 205) fluxes increased by 28, 25, 28, and 27% when compared to prior to fertilization under 24, 49, 98, and 198 kg N ha⁻¹ yr⁻¹, respectively. Fluxes in the third week after N fertilization (d 219) declined to 414, 479, 505, and 561 kg CO₂ ha⁻¹ d⁻¹ under 24, 49, 98, and 198 kg N ha⁻¹ yr⁻¹, respectively. Results are similar to those obtained in a study

conducted to compare greenhouse gas emissions under different N fertilization rates, which revealed greater CO₂ fluxes for up to 2 weeks after N application (Zhang et al., 2013).

Results averaged over the entire study showed strong quadratic correlation between CO₂ flux and N rate ($r^2 = 0.99^{**}$, Figure 16). Carbon dioxide flux during the study period increased from 292 ± 12 to 394 ± 13 kg ha⁻¹ d⁻¹ as N rate increased from 24 to 196 kg N ha⁻¹, respectively. This is similar to results obtained from a study conducted at the Bayinbuluk alpine grassland located in the southern Tianshan Mountains of Central Asia, with mean annual temperature -4.8°C (Li et al., 2012), which found that CO₂ emission significantly increased from 68 to 83 kg ha⁻¹ d⁻¹ with increasing N rate (NH₄NO₃) from zero to 150 kg N ha⁻¹, respectively (Li et al., 2012).

Seasonal CO₂ Fluxes:

June to August (Summer)

At every N rate, soil CO₂ flux was significantly correlated with N rate during the summer of 2012 ($r^2=0.98^{**}$, Figure 17) and 2013 ($r^2=0.99^{**}$, Figure 18). In summer 2012, average fluxes of CO₂ were estimated to be 400 ± 23 , 462 ± 17 , 500 ± 20 and 544 ± 23 kg CO₂ ha⁻¹ d⁻¹ under 24, 49, 98, and 198 kg N ha⁻¹ yr⁻¹, respectively. In 2012, highest fluxes during summer occurred on day 226, with an estimated average of 646 ± 30 , 677 ± 33 , 780 ± 14 and 813 ± 37 kg CO₂ ha⁻¹ day⁻¹ under 24, 49, 98, and 198 kg N ha⁻¹ yr⁻¹, respectively. This occurred two days after the addition of N, likely a factor of stimulated microbial activity due to N application. Additionally, soil temperature was greater than 30 °C and moisture content was higher than on any other sampling date, with an average value of 31.3%. In 2013, highest CO₂ flux was observed on d 135 (three days after N

application) with CO₂ fluxes of 470 ± 40, 548 ± 24, 566 ± 31 and 593 ± 12 kg CO₂ ha⁻¹ d⁻¹ under 24, 49, 98, and 198 kg N ha⁻¹ yr⁻¹, respectively. Those findings are similar to results reported by Mielnick and Dugas (2000), who found CO₂ flux as great as 587 kg CO₂ ha⁻¹ d⁻¹ in a tallgrass prairie at the Blackland Research Center, Temple, TX, USA in a Houston Black clay soil.

Summer emissions of CO₂ in 2013 were lower than in 2012, which may have resulted as an interaction of environmental factors including, but not limited to, variation in soil temperature and moisture content during the entire study (Figures 17 and 18). For example, soil moisture influences soil CO₂ flux rates by increasing and/or decreasing root respiration and/or microbial decomposition rates. Low moisture levels affect root respiration (Chen et al., 2000) and soil respiration rates by limiting the microbial activity of soil (Orchard and Cook, 1983).

September to November (Fall)

During the fall season, N rate was significantly correlated with soil CO₂ flux during 2012 ($r^2=0.98^{**}$, Figure 17) and 2013 ($r^2=0.99^{**}$, Figure 18). Fluxes of CO₂ ranged from 390 (24 kg N ha⁻¹) to 495 (196 kg N ha⁻¹) kg CO₂ ha⁻¹ d⁻¹, then declined by the end of November in both years after soil temperature decreased (Figures 19).

Nitrogen application at d 250 and 256 in 2012, and 2013, respectively did not increase fluxes compared to those observed during summer of both years, probably because of lower fall temperatures. Application of N at a higher rate produced greater cumulative emissions than at lower rates in the fall of both years. However, CO₂ flux increased in all N rates on d 249 (2012) after 65 mm of rainfall had fallen.

In general, the greatest increase in CO₂ fluxes among treatments was observed under 196 kg N ha⁻¹, probably because of the impacts of N fertilization on stimulation of microbial and root activities. The effect of soil temperature can often be observed in the CO₂ data. For example, in 2012, fluxes on d 291 and 298 were lower than the measured fluxes on d 260, likely a result of soil cooling from 21.2°C on d 290 to 12.7°C by d 305. In 2013, CO₂ fluxes were lowest at d 331, with estimated values of 89 ± 7, 100 ± 4, 111 ± 5 and 118 ± 2 kg CO₂ ha⁻¹ d⁻¹ under 24, 49, 98, and 198 kg N ha⁻¹ yr⁻¹, respectively. These low CO₂ fluxes were likely a result of bermudagrass dormancy and lower soil temperature (5.6 °C) during measurements (Esmaili and Salehi, 2012).

December to February (Winter)

In December of both years, significant correlation between CO₂ flux and N rate existed in both years. Fluxes were lower; likely a result of soil temperatures below 15°C at all measurement dates (Figure 19). Various studies have reported declining CO₂ fluxes during the winter months in different ecosystems, including forest ecosystems (Winston et al., 1997) and in row crops such as maize (Zang et al., 2013) and wheat (Billesbach et al., 2014).

Carbon dioxide fluxes on d 361(2012) had lowest values during the entire study with estimated values of 59 ± 5, 61 ± 1, 65 ± 14 and 71 ± 7 kg CO₂ ha⁻¹ d⁻¹ under 24, 49, 98, and 198 kg N ha⁻¹ yr⁻¹, respectively. Greatest values measured during the winter season were observed on d 240 (2012) with measured CO₂ fluxes of 128 ± 9, 149 ± 2, 154 ± 4, and 167 ± 8 kg CO₂ ha⁻¹ d⁻¹ under 24, 49, 98, and 198 kg N ha⁻¹ yr⁻¹, respectively. Greater CO₂ fluxes among treatments may have been related to an increase

in soil moisture (17%), and increases in soil temperature to 16.9°C on d 340, in 2012 (Figure 20). However, greater CO₂ emissions were not observed during the same period of 2013 even though soil moisture was higher in 2013 than in 2012. This may indicate that temperature, rather than moisture, is the controlling weather variable in this study. In support of that speculation, significant changes in CO₂ fluxes were observed within the same treatment during the entire study (Tables 13 and 14). Recently, Zhang et al., (2014) confirmed that soil temperature at 10 cm soil depth significantly influenced soil CO₂ flux.

Fluxes from March to May (Spring)

Greater CO₂ fluxes among treatments on d 151 (2012) was most likely caused by a combination of an average 2.7°C and 3.7 % increase in soil temperature and moisture content, respectively, which improved N uptake and increased microbial activities. Highest spring CO₂ fluxes (2013) were observed on day 149 (21.1°C soil temperature and 31.9% moisture content) with estimated values of 507, 567, 613 and 677 kg CO₂ ha⁻¹ d⁻¹ under 24, 49, 98, and 198 kg N ha⁻¹ yr⁻¹, respectively.

Annual Cumulative Fluxes

Cumulative emission of CO₂ in 2012 was greater than measured in 2013. Cumulative emissions of CO₂ increased rapidly after several N applications, especially during summer, reflecting peaks in fluxes after fertilization. Cumulative emissions of CO₂ in 2012 were estimated at 121 ± 4.4, 135 ± 4.4, 146 ± 4.8, and 159 ± 5.5 compared to 93 ± 3.5, 103 ± 2.7, 115 ± 4.5, and 129 ± 3.3 Mg CO₂ ha⁻¹ y⁻¹ under 24, 49, 98 198 kg N ha⁻¹ yr⁻¹, respectively (Figure 21). Similarly, Allaire et al., (2008) measured CO₂

emissions from four different levels of lawn management. The four lawn management approaches used on the mixed cool-season lawn were: 1) N fertilizer with clipping removal and frequent mowing, 2) no fertilizer with clippings returned and frequent mowing, 3) treatment #2 with only three mowings, and, 4) treatment #2 with only 1 mowing. They measured CO₂ emissions from May 11 to November 15 at 7 to 14 d intervals for one year. Mowing frequency had a greater impact on CO₂ flux than fertilizer. Frequently mowed lawns had an accumulative CO₂ emission of 100 Mg ha⁻¹ year⁻¹, which was four times higher than in lawns mowed 3 or 1 times. In addition, at a high mowing frequency clipping removal did not affect CO₂ emissions (Allaire et al., 2008). In our work clippings were not removed. This continual re-introduction of N rich plant tissue would likely further stimulate CO₂ fluxes.

In contrast, in a study conducted by Zhang et al., (2014) in a paddy rice field, CO₂ fluxes were measured June 9 to December 23 in 2011 using a static chamber method under 3 N rates of 150, 210, and 300 kg N (urea) ha⁻¹. Measured CO₂ flux did not show significant differences with increasing N rate from 150 (88 Mg ha⁻¹ yr⁻¹) to 300 (90 Mg ha⁻¹ yr⁻¹) kg N ha⁻¹ (Zhang et al., 2014).

Conclusions

Average daily fluxes of CO₂ for the entire study were significantly ($r^2=0.99^{**}$) correlated with N rate with estimated values ranging from 292±12 to 394±13 kg CO₂ ha⁻¹ day⁻¹ under 24 and 198 kg N ha⁻¹ yr⁻¹, respectively. These results indicate that CO₂ emissions significantly increased as N rate increased. In addition, fertilization during higher soil temperatures and moisture content resulted in larger fluxes of CO₂. Data

suggest that lower N application rate such as $49 \text{ kg N ha}^{-1} \text{ yr}^{-1}$ may be a better choice for mitigation of CO_2 emissions under bermudagrass management. Further research may be required to determine whether additional factors may contribute to differences in CO_2 emissions among treatments such as root activity and soil NO_3^- and NH_4^+ concentrations and fates. Further research is needed to investigate CO_2 emissions among most common warm season turfgrasses such as centipedegrass and zoysiagrass under different geographical areas with different climate regimes to determine suitable, regionally appropriate species for reducing CO_2 emissions.

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Table 11. Significance of N rates (kg ha⁻¹) on CO₂ flux, pairwise multiple comparisons among N rates calculated at $\alpha \leq 0.05$. Data shown is averaged over 2012.

N rate (kg ha ⁻¹)	N rate (kg ha ⁻¹)	Estimate	DF	t Value	Pr > t	Adj P
24	49	-39.77	3	-1.03	0.3041	0.7316
24	98	-69.34	3	-1.80	0.0741	0.2783
24	196	-103.02	3	-2.67	0.0083	0.0411
49	98	-29.57	3	-0.77	0.4445	0.8695
49	196	-63.25	3	-1.64	0.1030	0.3594
98	196	-33.68	3	-0.87	0.3839	0.8188

Table 12. Significance of N rates (kg ha^{-1}) on CO_2 flux, pairwise multiple comparisons among N rates calculated at $\alpha \leq 0.05$. Data shown is averaged over 2013.

N rate (kg ha^{-1})	N rate (kg ha^{-1})	Estimate	DF	t Value	Pr > t	Adj P
24	49	-27.98	3	-0.73	0.4634	0.8831
24	98	-61.26	3	-1.61	0.1093	0.3762
24	196	-100.4	3	-2.64	0.0090	0.0444
49	98	-33.28	3	-0.87	0.3832	0.8184
49	196	-72.42	3	-1.90	0.0587	0.2308
98	196	-39.13	3	-1.03	0.3054	0.7336

Table 13. Significance of CO₂ flux within treatments over time. Data shown is averaged over 2012.

N rate (kg ha ⁻¹)	Estimate	DF	t Value	Pr > t
24	323.27	3	11.85	<0.0001
49	363.05	3	13.31	<0.0001
98	392.61	3	14.39	<0.0001
196	426.30	3	15.63	<0.0001

Table 14. Significance of CO₂ flux within treatments over time. Data shown is averaged over 2013.

N rate (kg ha ⁻¹)	Estimate	DF	t Value	Pr > t
24	254.39	3	9.45	<0.0001
49	282.37	3	10.48	<0.0001
98	315.66	3	11.72	<0.0001
196	354.79	3	13.17	<0.0001

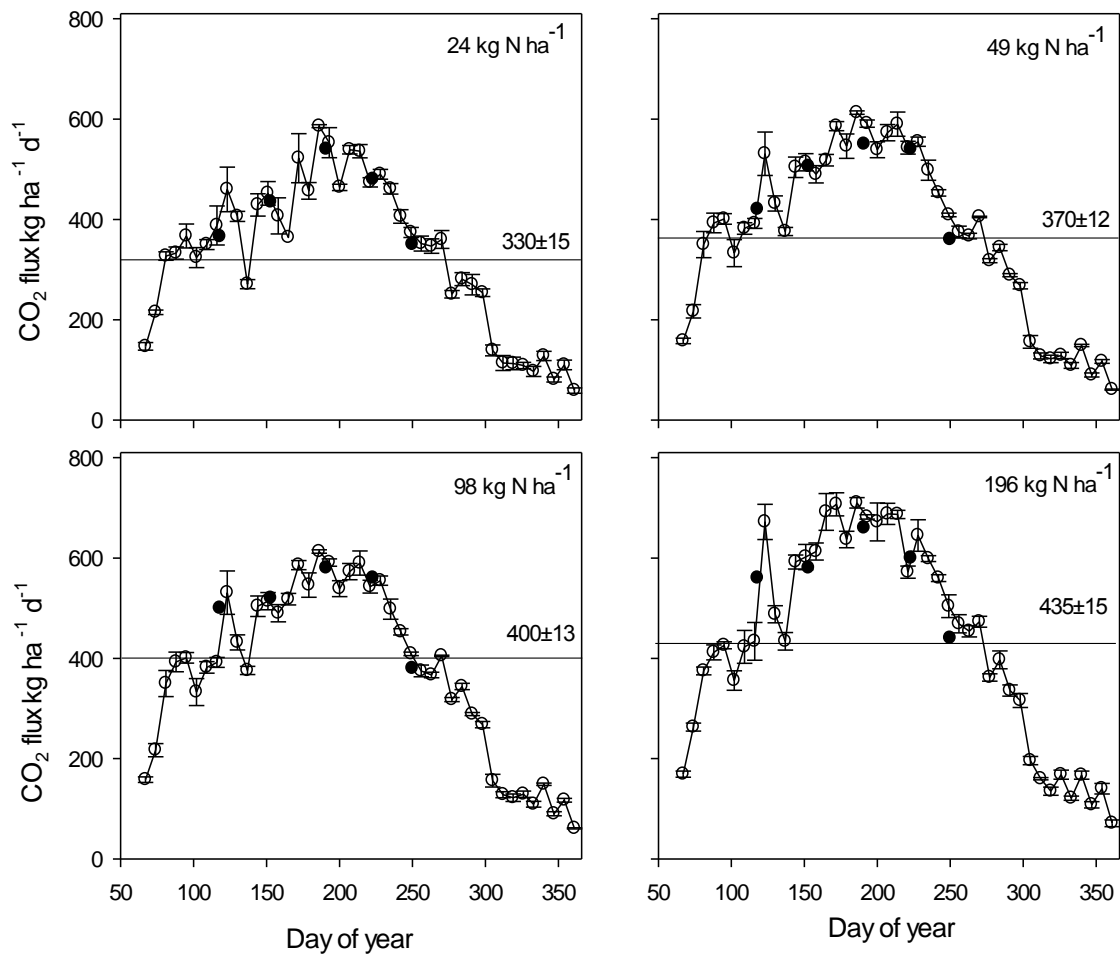


Figure 12. Carbon dioxide flux as affected by nitrogen (N) rate during 2012. Solid line is the mean±standard error of 43 weeks. Filled circles indicate fertilizer application dates.

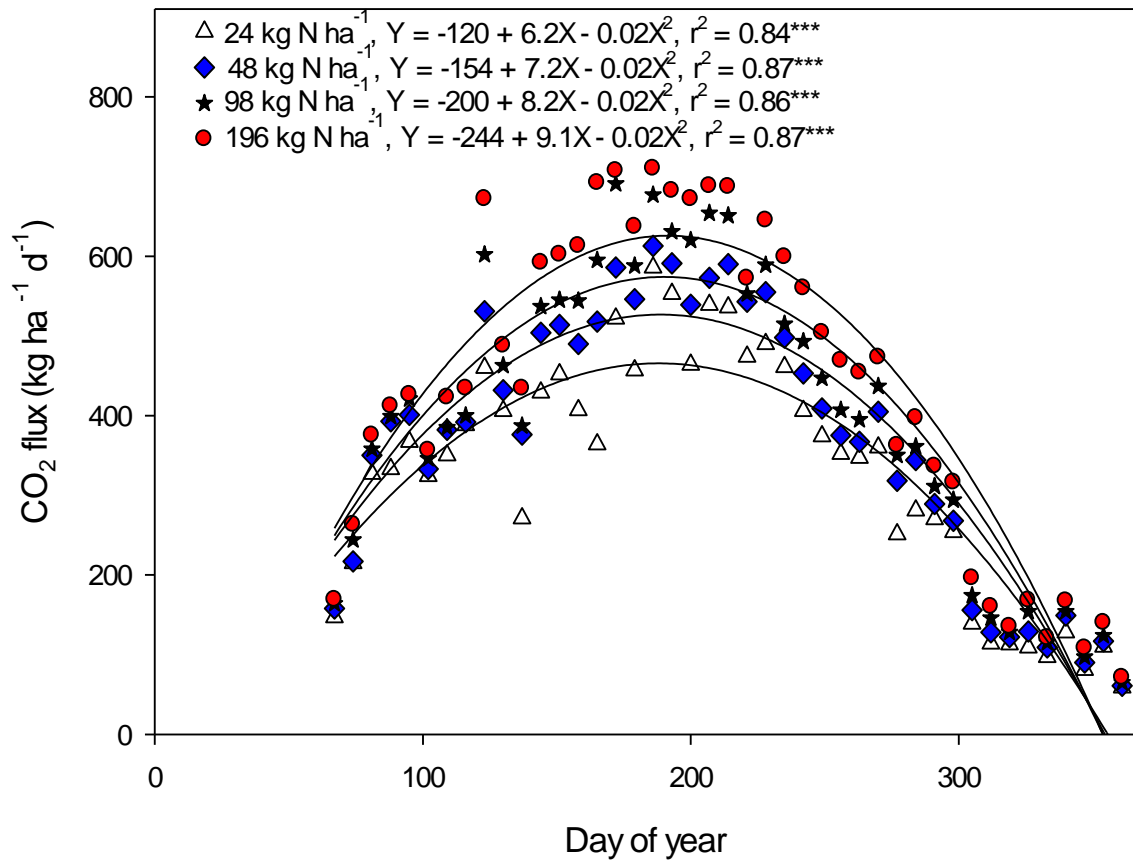


Figure 13. Carbon dioxide flux as affected by N rate, 2012 with quadratic fitted curves for each N rate over time. Equations are describing the pattern of each N rate over time. All curves fit at r^2 ranged from 0.84 to 0.87.

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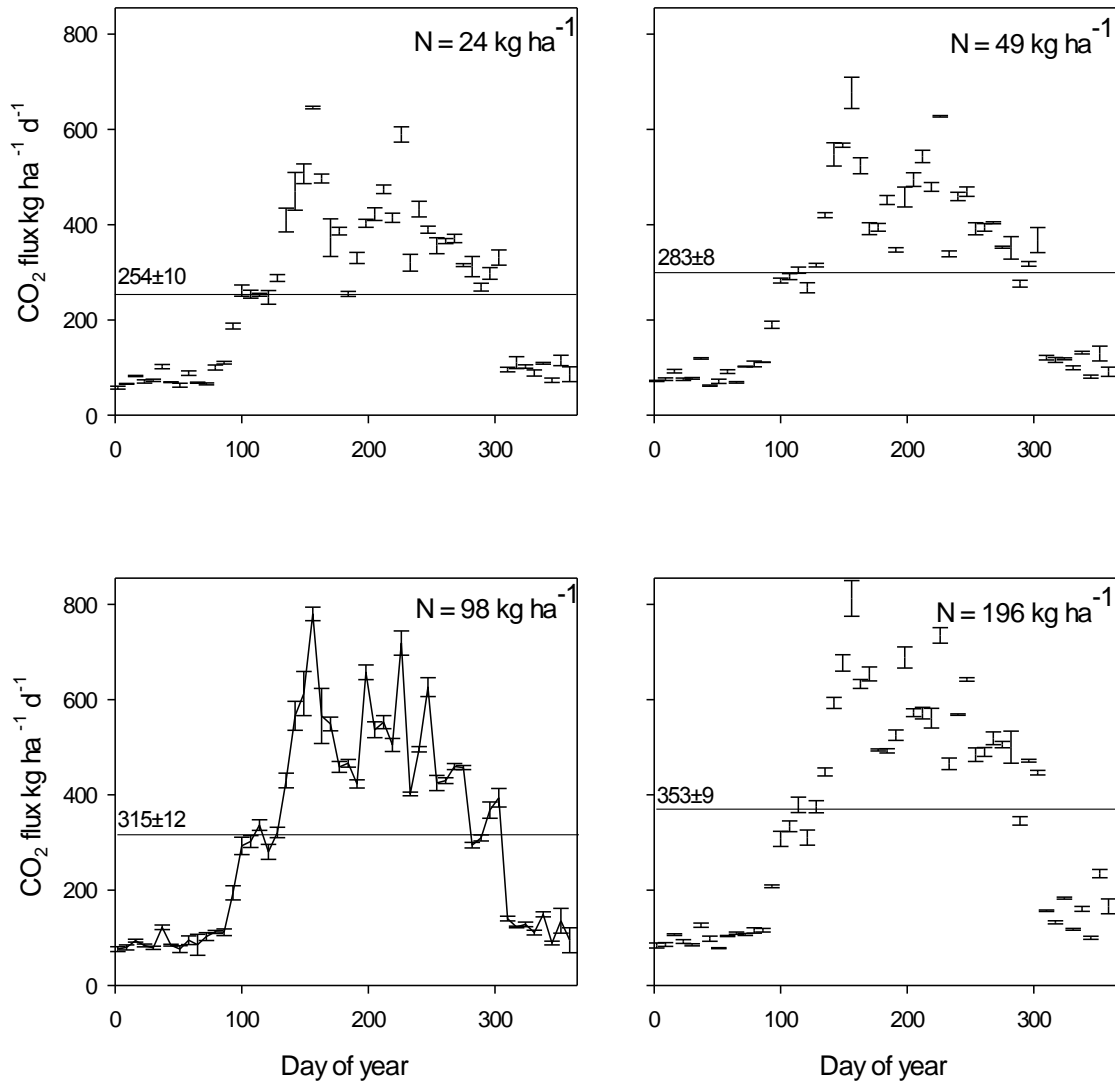


Figure 14. Carbon dioxide flux as affected by nitrogen (N) rate during 2013. Solid line is the mean+standard error of 52 weeks of data collection. Filled circles indicate N fertilizer application dates.

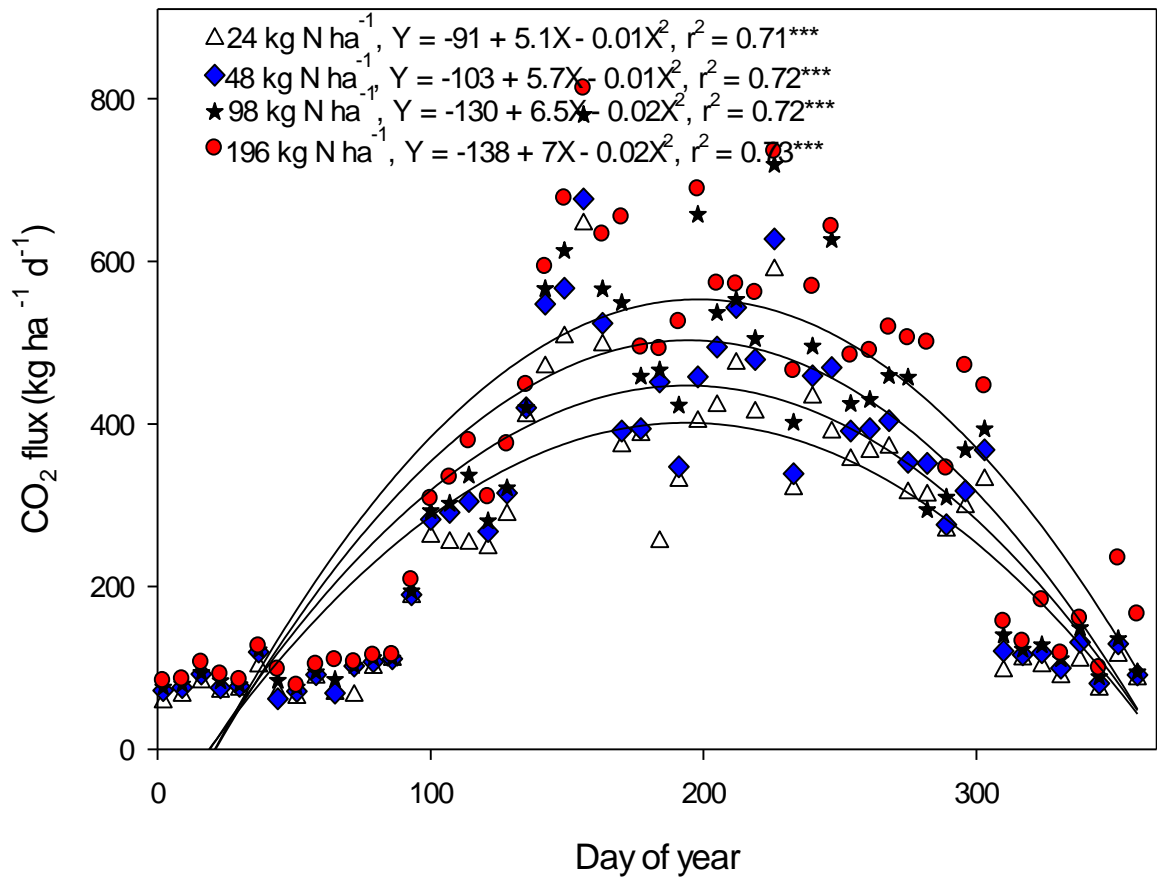


Figure 15. Carbon dioxide flux as affected by N rate, 2013 with quadratic fitted curves for each N rate over time. equations are describing the pattern of each N rate over time. All curves fit at r2 ranged from 0.71 to 0.73.

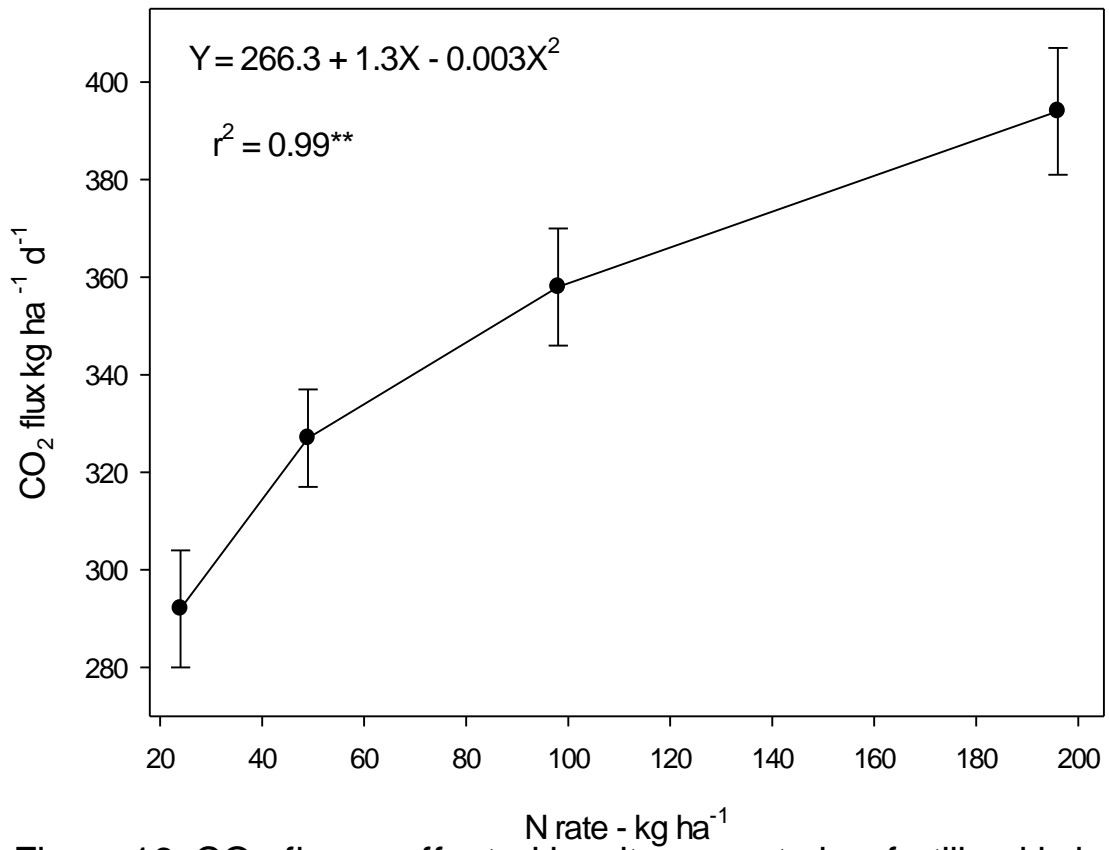


Figure 16. CO₂ flux as affected by nitrogen rate in a fertilized hybrid bermudagrass fairway. Bars represent standard errors about the mean.

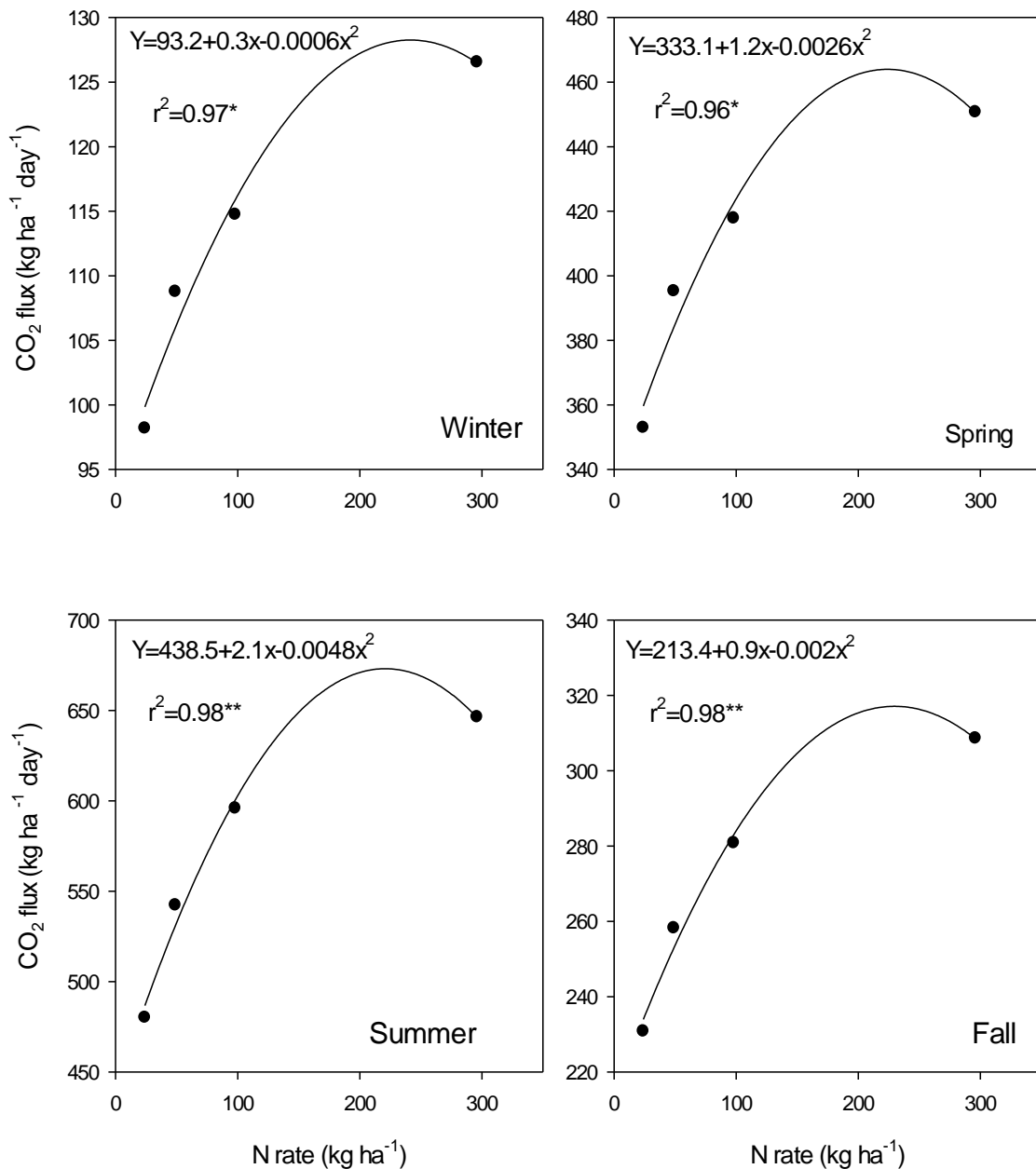


Figure 17. The relationship between N rate and CO₂ flux during different seasons in 2012, fertilized hybrid bermudagrass with quadratic curves fitted for each season. All curves fit at r² from 0.96 to 0.98.

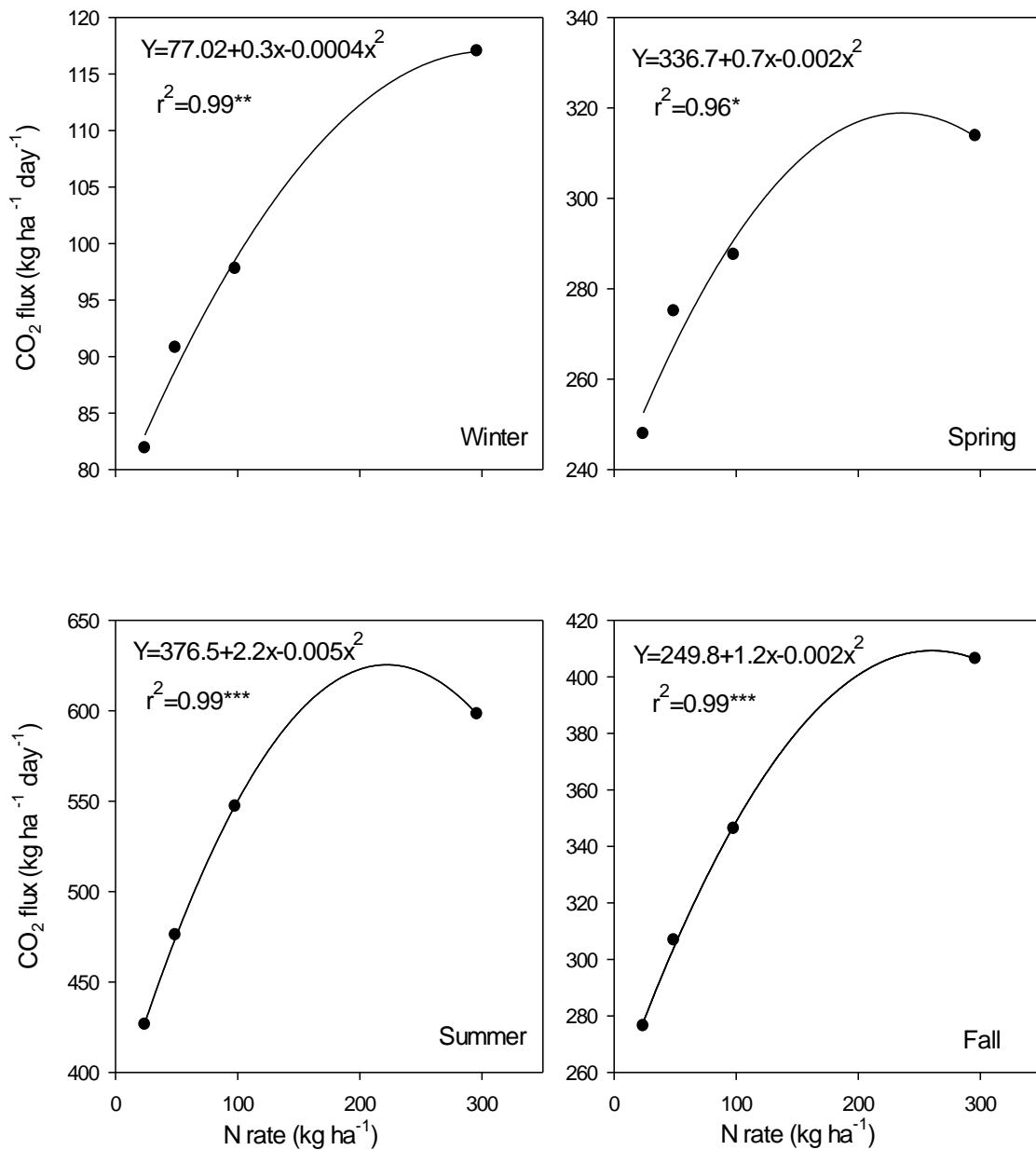


Figure 18. The relationship between N rate and CO₂ flux during different seasons in 2013, fertilized hybrid bermudagrass with quadratic curves fitted for each season. All curves fit at r² from 0.96 to 0.99.

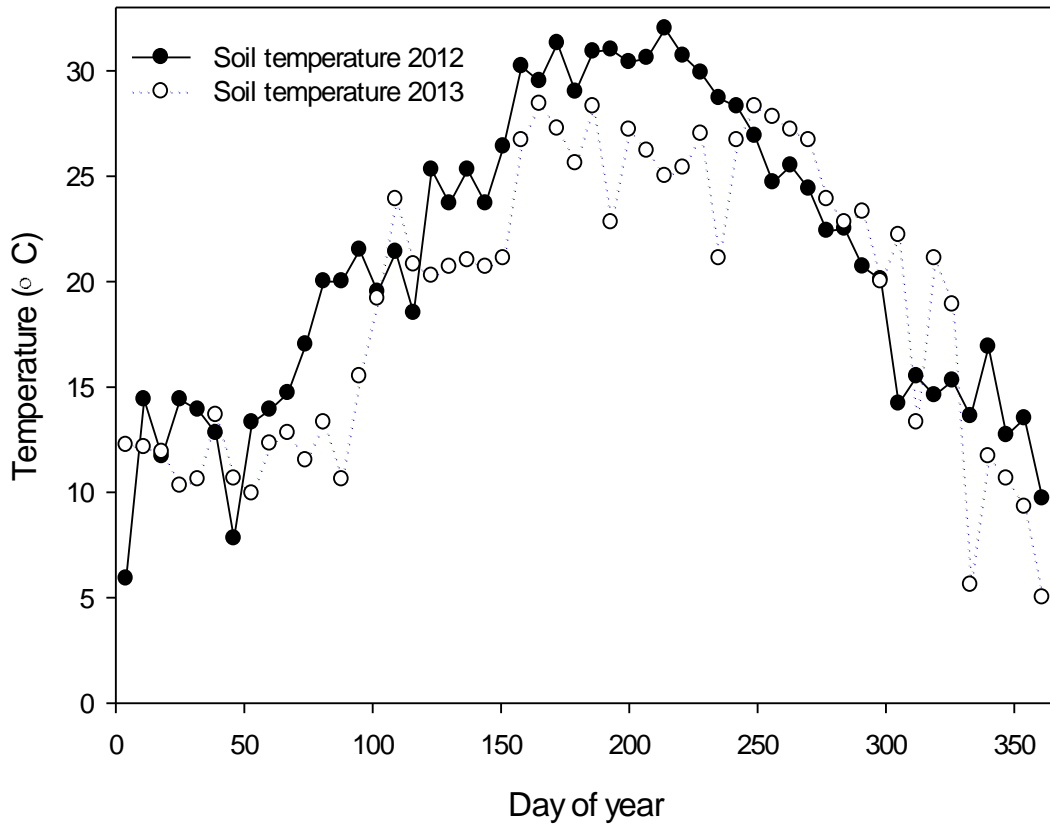


Figure 19. Average soil temperature for each day in 2012 and 2013 at a 10 cm soil depth, Auburn, AL.

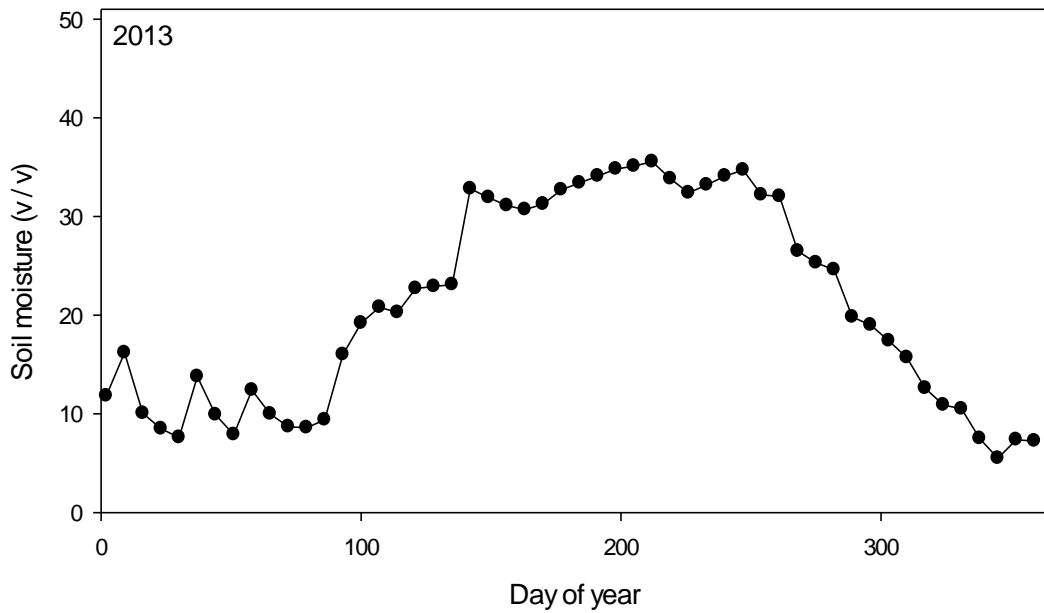
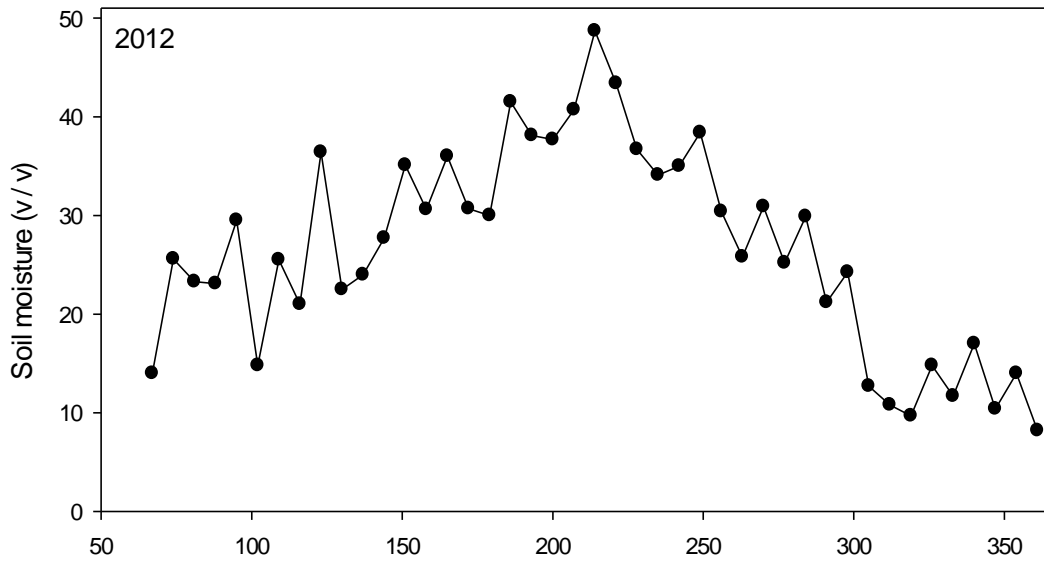


Figure 20. Average soil moisture content (%) at a 10 cm depth, 2012 and 2013, Auburn, AL.

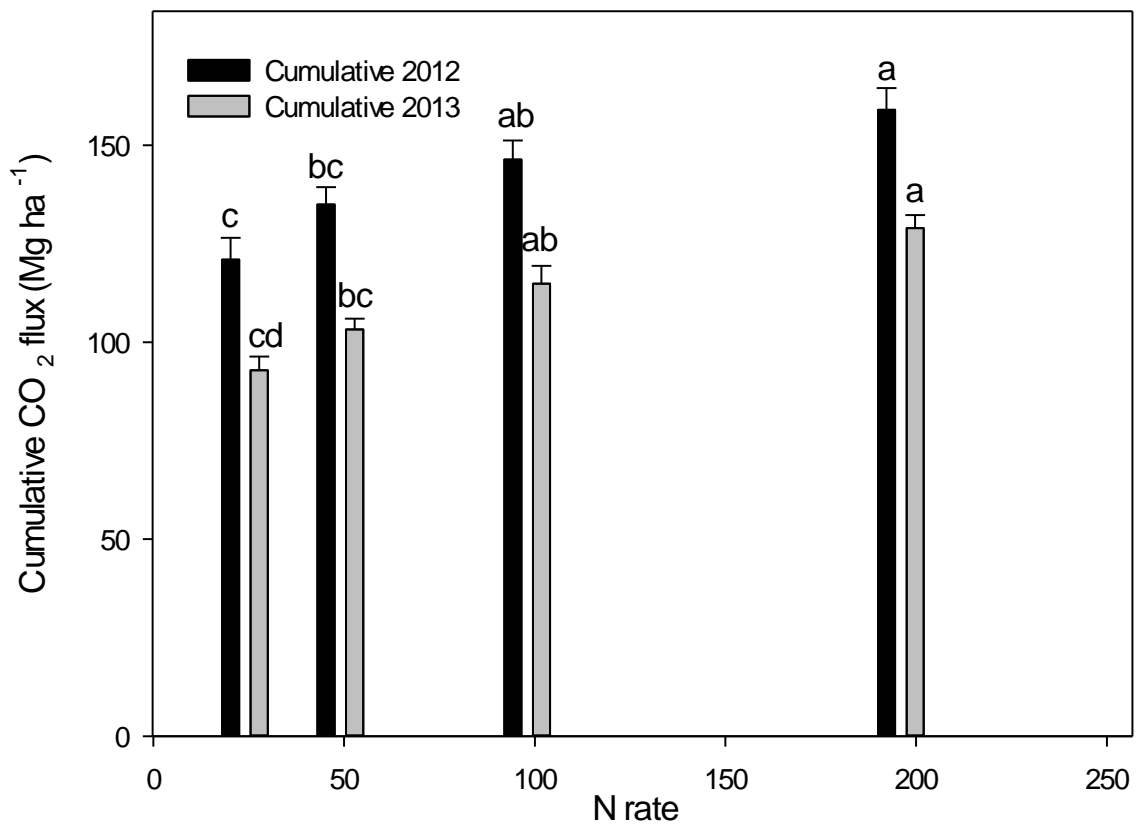


Figure 21. Cumulative CO₂ flux as affected by N rate during 2012 and 2013. Bars with similar letters within each N rate indicate no significant difference between those means. Bars represent standard error about the mean.

CHAPTER IV

Decomposition, and Carbon and Nitrogen Release from Turfgrass

Abstract

Turfgrass cover in the U.S. is expanding because of increasing urbanization and the addition of approximately 675,000 ha of residential property every year. Such perennial grass covers could have a significant effect on soil carbon (C) and nitrogen (N) cycling. Despite its large-scale presence in the urban ecosystem, the role of turfgrasses in C and N cycling in urban environments in the southeastern U.S. has not been documented, and studies addressing C and N cycling in warm and cool season turfgrass in the southeastern United States are lacking. The objective of this study was to determine decomposition rates and C and N release under warm and cool season turfgrass. The study was initiated in May 17, 2012 and conducted for 46 weeks at the Auburn University Turfgrass Research Unit. Litter from five turfgrasses was selected for this study, including: bermudagrass (*Cynodon dactylon* (L.) Pers. *C. transvaalensis* Burt & Davy), centipedegrass (*Erechtholmoa ophioides* Munroe Hack.), St. Augustinegrass (*Stenotaphrum secundatum* L.), tall fescue (*Lolium arundinaceum* S.J. Darbyshire), and zoysiagrass (*Zoysia japonica* Stued.). Litter was placed into nylon bags measuring 10 × 20 cm with 50 to 60 μm openings based on an oven dry rate of 3.6 Mg ha⁻¹. Litter bags were retrieved from the field after 0, 1, 2, 4, 8, 16, 24, 32, and 46 weeks, and contained

litter analyzed for total C and N. A double, four-parameter exponential decay model was used to describe mass, C, and N loss. Results indicated that tall fescue decomposition occurred more rapidly compared to warm season turfgrasses.

Introduction

Understanding litter decomposition process in a given ecosystem is vital due to its effect on greenhouse gas concentration and biogeochemical cycling in terrestrial ecosystems. During decomposition significant amounts of greenhouse gas including CO₂, CH₄, and N₂O are released (Berg and McClaugherty, 2008). Litter decomposition plays an important role in global C and nutrient cycling (Attiwill and Adams, 1993; Lambers et al., 1998). Plants may also play an important role in C and nutrient cycling through the quality and quantity of their residues (Aerts and Chapin, 2000; Swift et al., 1979; Rovira and Vallejo, 2007). The quality and the type of litter material influence soil organic matter content (Wardle et al., 2002). Increasing and/or decreasing organic matter in soil can influence cation exchange capacity (CEC), water infiltration rate, aggregation stability, and soil quality.

Organic matter with higher C:N and/or lignin:N ratios decompose slower, which reflect lower rates of N mineralization and increase N immobilization in microbial biomass (Adams and Attiwill, 1986). Moreover, the ability of soil microorganisms to decompose and/or mineralize organic matter depends on the chemical structure of the C compounds (Krauss et al., 2004). Complex C compounds such as lignin can retard litter decomposition. Thus, the composition of plant residues, in particular C, N, and lignin concentrations, determines the rate and extent of decomposition of such residues. With

slow decomposition owing to high C:N ratio and lignin concentrations in turf litter, retention of C in turfgrass systems increases with time until a quasi-equilibrium is established within the bounds of soil and climatic controls.

Lignin is an aromatic polymer derived from phenylalanine. It is a high molecular weight polyphenolic compound and is one of the most recalcitrant fractions of plant residue (Stubbs et al., 2009). It is a vital element of plant secondary cell walls and comprises a large fraction of plant litter, with estimated values ranged from 19.1 in *Lithocarpus dealbata* to 35.5% in *Schima wallichii* (Devi and Yadava, 2010). Its concentration has been widely used as an index of organic matter quality. Its concentration alone, and/or lignin:N content can be used as an indicator of decomposition rates (Sterjiades and Erikson, 1993; Hobbie, 1996; Kitayama et al., 2004).

Litter with higher initial lignin contents has slower decomposition rates. For example, the decomposition of *Schima wallichii* litter had a slower decomposition rate than *Lithocarpus dealbata* (Devi and Yadava, 2010). However, concentrations of the lignin fraction increased as decomposition proceeded (Berg, 2000) and the litter became enriched with lignin (Devi and Yadava, 2007). Earlier work showed that as lignin concentration increased decomposition rate was suppressed (Fogel and Cromack, 1977).

Litter decomposition occurs in two stages. First, decomposition of a labile portion including sugars, starch, soluble and unprotected cellulose and hemicelluloses takes place by soil microbes, in a process controlled by climate (Aerts, 1997). In the second stage, the influence of climate on decomposition gradually decreases to zero (Vargas et al., 2006), and the decomposition of recalcitrant substances, including cellulose and lignin

(Wieder and Lang, 1982), become more dominant in the residue. In some cases the rate of decomposition approaches zero. This second portion is slowly decomposed, contributing to the development of SOM. Each molecule of SOM contains 58% C; however, organic C content can vary considerably with molecular composition (Stevenson and Cole, 1999).

Organic C comprises three pools (Wander, 2004). First is the active pool that consists of substances such as carbohydrates that are easily broken down via a vast range of microorganisms, with turnover times estimated to be ranged from months to a few years (Derenne and Largeau, 2001). Second, the slow pool, which represents stabilized decomposition products such as lignin, with turnover times from decades to hundreds of years (Coleman and Jenkinson, 1996). Third is the passive pool, which includes recalcitrant SOC such as charcoal, with estimated residence times to from 5000 to 10000 years (Skjemstad et al., 2002; Swift, 2001). In the slow and recalcitrant pools, SOC is stabilized due to physical protection and chemical resistance against microbial degradation. Physical protection is provided by physico-chemical interaction of the soil mineral texture and organic material. Physical stabilization is due to aggregate formation and structural stability of the organic molecule. These two mechanisms are not independent but interact with each other (Krull et al., 2003). The aggregate provides physical protection to the SOC for a long time, unless disrupted and exposed to physical factors such as temperature and moisture that allow for further microbial activity. Krull et al. (2003) suggested that chemical recalcitrance may be the only mechanism for SOC protection over long time periods.

Chemical recalcitrance is achieved during the process of humification (Basler, 2005), which is a part of decomposition that leads to formation of stable humic substances (HS). In addition, HS are important in determination of soil properties such as water holding capacity, soil structure, aggregation, and nutrient retention. Humic substance formation and turnover in soil is governed by microorganisms (Basler, 2005). Humification plays an important role in the process of SOC sequestration. The chemical perspective on the role of microbes in humus formation suggests the following pathways of HS formation (Basler, 2005): (1) degradation of plant material and abiotic condensation of the products to form humic substances, (2) degradation of primary resources and re-synthesis into recalcitrant components in the bodies of microbes which remain in the soil after their death, (3) selective preservation wherein easily degradable plant material is degraded and the large resistant structures form humic substances, and (4) microbes participate directly in humus formation by enzymatic activities. Therefore, from a microbiological perspective, humification occurs in three phases (Basler, 2005): 1) rapid initial decomposition of primary plant residues which is performed by a wide range of microorganisms; 2) slow decomposition of primary plant structural components; in this step the degradation rate slows because labile C has been used by microorganisms, and larger plant structures such as lignin, cellulose and hemicelluloses remain. These plant structures are larger in size than labile C from phase 1 and must be decomposed outside the cell prior to uptake into the cell for degradation; and 3) alteration of SOC and HS genesis - in this phase microbial activity oxidizes and alters HS and/or generates and polymerizes aromatic compounds to form HS. Since humification is controlled by soil microbes, the same factors affecting soil microbes also affect the process of HS

formation. In case of climatic conditions that do not favor the growth of microorganisms the plant residue will be stored in the soil as SOC (Balsler, 2005).

The SOC can be protected as long as there is no soil disturbance. Once a disturbance occurs SOC is exposed to microbial degradation, which may or may not lead to HS formation depending on the structure of the microbial community. For example, fungi can degrade plant residue more efficiently than bacteria because fungal cell walls are highly recalcitrant, and release more C as CO₂ (Balsler, 2005). Actinomycetes play a major role in the formation of HS in soil by decomposing plant and insect residue such as cellulose, hemicellulose, lignin, chitin (Nelson, 1997). Thus, humus formation is largely controlled by microorganisms.

Although decomposition processes have been studied in row crops and forest settings, there is limited research which examines decomposition rate of C and N dynamics in the long-term, non-tilled that are warm and cool season turfgrasses. Thus, there is a need to evaluate C and N dynamics in warm and cool season turfgrasses in the southeast United States. The objective of this study was to assess mass loss and C and N decomposition rates from turfgrass clippings under field conditions.

Materials and Methods

Site Description and Sampling

A field decomposition study was initiated in May 17, 2012 and conducted for 46 weeks at the Auburn University Turfgrass Research Unit (TGRU) (32.58° N, 85.50° W) on a Marvyn loamy sand soil (fine-loamy, kaolinitic, thermic Typic Kanhapludult). The

mean annual air and soil temperatures were 17 and 19°C, respectively and the mean annual precipitation was 1233 mm (2011-2014, AWIS, 2014).

Prior to establishment of the experimental plots, turfgrasses were maintained at appropriate mowing heights for each grass. Centipedegrass, St. Augustinegrass, and zoysiagrass were maintained 3.8 cm, compared to 1.9 and 6.4 cm for bermudagrass and tall fescue, respectively. Supplemental irrigation was provided in the absence of rainfall so that irrigation + precipitation equaled 2.5 cm wk⁻¹. All turfgrasses were harvested at 2 cm above soil level using a push lawn mower. Litter was transported immediately to the laboratory. Turfgrass clippings were harvested from bermudagrass, centipedegrass, St. Augustinegrass, tall fescue, and zoysiagrass. Litter of each grass was mixed separately, and weeds or other debris were removed. Litter was placed into nylon bags measuring 10 × 20 cm with 50 to 60 µm openings rate at 30 g an oven dry bag⁻¹ (3.6 Mg ha⁻¹). Thirty six litter bags were placed directly into the thatch layer (the area had been trimmed free of verdure) on the same grass. The experimental design was a randomized complete block design with five treatments (bermudagrass; centipedegrass; St. Augustine grass; tall fescue; and zoysiagrass) examining their effect on biomass decomposition, and C and N release. The treatments were replicated four times. Four steel staples were used to secure corners of the sealed litterbag to the thatch layer.

Bags were retrieved (4 bags/grass) from the field after 0, 1, 2, 4, 8, 16, 24, 32, and 46 weeks. Retrieved bags were emptied into plastic containers and oven-dried at 55°C for 72 hours and weighed for dry-matter determination. Litter was ground to pass a 1 mm-sieve and analyzed for total C and N using LECO TruSpec CN analyzer (Leco Corp, St. Joseph, MI). All data were converted to an ash-free dry weight (AFDW) basis by ashing

1 g of sample in a muffle furnace at 450°C for 16 hours (Cochran, 1991). Initial values of lignin concentration in leaf litter were assessed by the acid-detergent digestion technique (Sluiter et al., 2011). Acid detergent fiber technique measures cellulose, lignin, and ash, and represents the insoluble components of cell walls. Acid detergent lignin (ADL) is an estimate of the lignin content. Neutral detergent fiber (NDF) comprises the insoluble components of cell walls (cellulose, hemicellulose, and lignin). Acid detergent fiber (ADF) is a measure of cellulose and lignin (Stubbs et al., 2009). Soil moisture and temperature were measured on weekly basis using an auxiliary sensor.

A double, four-parameter exponential decay model (Hunt, 1977; Weider and Lang, 1982) was used to describe the decay pattern as following:

$$Y = Ae^{-bx} + Ce^{-dx},$$

Where Y = remaining mass (normalized %),

A = the labile portion,

C = the recalcitrant portion,

b and d are the labile and the recalcitrant constants, respectively, and

x = time in weeks.

Means, standard errors, and statistical significance of treatments were determined at the 95% confidence level using mixed models procedures within PROC MIXED (SAS Institute, 2007). Least squares estimates for nonlinear models were determined using four parameter double exponential decay models (Systat Software, 2009). Double exponential decay models were employed as the basis for comparison of mass, N, and C loss between dates among all grasses. Multiple comparisons were performed by Fisher LSD test.

Results and Discussion

Initial Fiber Analysis

Fiber composition of harvested clippings had significant variation among turfgrasses. For example, on a dry matter basis (Figure 22), St. Augustinegrass litter was significantly higher in acid detergent fiber (cellulose + lignin) (ADF) and ash contents when compared to bermudagrass, centipedegrass, tall fescue, and zoysiagrass. Zoysiagrass displayed significantly higher lignin concentration in litter, $6 \pm 0.3\%$, while tall fescue had the lowest content, at 3 ± 0.3 .

The initial harvest of bermudagrass clippings had significantly higher neutral detergent fiber (hemicellulose + cellulose + lignin (approximately total cell wall)) (NDF) than found, measuring 85% (Figure 22). In contrast, tall fescue had lowest NDF, measured at 63%. Fiber contents play an important role in accelerating and/or suppressing decomposition processes. Berendse (1994) reported that higher lignin content can be used as indicator of long-lived leaves.

Double Exponential Decay Models

Analysis of variance indicated significant differences among turfgrasses for decay over time (Table 15). All regression equations were significant ($p < 0.0001$). Table 15 presents residue persistence normalized to 100% of initial oven dry of ash free weight. Normalized equations suggest an approximation of labile (A) and recalcitrant (C) litter on a percent basis. The coefficients b and d estimate the decay of labile and recalcitrant portions, respectively (Hunt, 1977; Weider and Lang, 1982). Differences in decay rates

are seen by comparing the b (labile constant) and d (recalcitrant constant) values from each equation (Hunt, 1977; Weider and Lang, 1982).

Remaining Mass

In all cases, the regression models were significant ($P < 0.0001$) with reasonably high adjusted R^2 values (Table 15). All data were expressed on a normalized basis (percent remaining) (Figure 23). Figure 23 indicates that the litter mass of tall fescue measured after 16 weeks decreased 84%, from an initial equivalent of 360 g m^{-2} to 58 g m^{-2} . Thereafter, litter decomposition of tall fescue was close to zero. In contrast, zoysiagrass had very slow decomposition, with only 25% loss from an initial equivalent of 360 g m^{-2} to 270 g m^{-2} . At the end of the study, final mass of zoysiagrass was decreased by 45%, to 197 g m^{-2} . The faster decomposition rate of tall fescue can be partially explained by the lower initial ADF content.

Time also was an important factor in the rate of litter loss (when litter loss was calculated). During the first 8 weeks, average litter loss was estimated at 16% in zoysiagrass, as compared to 23.8, 18.8, 24.2, and 44% in bermudagrass, centipedegrass, St. Augustinegrass, and tall fescue, respectively (Figure 23). After 16 weeks, average litter loss was estimated at 25.1% in zoysiagrass, compared to 47.5, 43.1, 73.9, and 44.5% in bermudagrass, centipedegrass, St. Augustinegrass, and tall fescue, respectively. After 46 weeks, average litter loss was estimated at 45.4% in zoysiagrass, compared to 61.7, 73.7, 72.2, and 86.8% in bermudagrass, centipedegrass, St. Augustinegrass, and tall fescue, respectively (Figure 23). Similar results were observed by Tripathi et al (2013). They found that in a one year decomposition study, combinations of litters of the leguminous tree *Colophospermum mopane* with *Cenchrus ciliaris* or *Lasiurus*

sindicus grasses decomposed rapidly during the first 4 month, a value of 33% of total mass (Tripathi et al., 2013).

The labile decay constant of zoysiagrass residue (0.12) was 2.6 times greater than that of centipedegrass (0.05) but similar to that of tall fescue (0.10). The effects of grass species on decay of recalcitrant portions were more distinct. The recalcitrant decay constant of zoysiagrass (0.0096) was more than 100 times greater than that of centipedegrass (8.28×10^{-19}) and tall fescue (8.12×10^{-19}) combined. Rapid decay of turfgrass tissues are typically related to warmer soil temperatures (Dang et al., 2009; Fernandes et al., 2009; Ferreira and Chauvet, 2011a, b; Geraldés et al., 2012) (Figure 24) and increased moisture contents (Fraser and Hockin, 2013) (Figures 24 and 25) which stimulate both soil microbial activity and arthropods involved in decomposition processes.

Labile portions of warm season turfgrass ranged from 16.9 to 89.5%, and recalcitrant portions were 84.1 and 13.8% under zoysiagrass and centipedegrass, respectively (Table 15). In contrast, tall fescue had a labile portion of 91.2%, and a recalcitrant portion of 9.5% (Table 15). Such contents could significantly influence decomposition processes by acceleration and/or suppression. Increasing the labile fraction is associated with decreasing the recalcitrant portion and will increase the decomposition rate.

Table 16 shows the calculated mass persistence comparison expressed at a 95% confidence interval after 46 weeks, for all grasses. There were significant differences among all selected turfgrasses. Tall fescue litter mass measured after 46 weeks decreased by 87% from an initial equivalent of 360 g m^{-2} to 48 g m^{-2} . In contrast, zoysiagrass

decreased from an initial equivalent of 360 g m^{-2} by 45% to 197 g m^{-2} . These differences may be due to fiber structures in addition to labile and recalcitrant portions.

Carbon Release from Turfgrass Clippings

All C data were expressed on a normalized basis (percent remaining) (Figure 26). In all cases, the regression models were significant ($P < 0.0001$) with reasonably high adjusted R^2 values (Table 15). As with the mass models, the C rate constant values c and d were larger for zoysiagrass residue than for centipedegrass, St. Augustinegrass, and tall fescue residues combined. Initial C concentrations of turfgrasses differed slightly, from 42.8% in zoysiagrass, compared to 40.6, 42.2, 41.7, and 39.1 under bermudagrass, centipedegrass, tall fescue, and St. Augustinegrass, respectively. After 16 weeks of field incubation, zoysiagrass had lost only 21.7% of C content, while tall fescue lost 84%. Carbon loss models were comparable to mass loss. This loss was attributed to mass lost through microbial respiration of C (as CO_2) to the atmosphere (Wood and Edwards, 1992).

Tall fescue C decreased faster during warmer temperatures. The labile decay constant of tall fescue (0.11) (Table 15) was greater than that of centipede and Saint Augustine grasses combined. The recalcitrant decay constant of tall fescue (1.07×10^{-17}) was nearly two times greater than centipedegrass (4.93×10^{-18}) and more than 100 times lower than zoysiagrass (0.01). The labile C portion of centipedegrass, tall fescue, and St. Augustinegrass was greater than 87%, and the recalcitrant portion of all these grasses was lower than 14%. In comparison, zoysiagrass had approximately 11% labile and 90% recalcitrant fractions. This may be the reason behind resistance to decay.

The time to decompose the turfgrass clipping varied with species. At 24 weeks, tall fescue C had declined 85.2%, from an initial equivalent of 360 g C m⁻² to 53 g C m⁻², while zoysiagrass decreased 25.7%, to 268 g C m⁻². These differences may have been due to the chemical fiber structure such as NDF in the clippings. After 46 weeks, there were significant differences in C among turfgrass clippings with exception of bermudagrass and centipedegrass (Table 17).

Table 18 illustrates the relationship between C concentration in clippings and turfgrass species. Carbon concentrations were significantly correlated with sampling time in bermudagrass ($r^2 = 0.78^{**}$) and St. Augustinegrass ($r^2 = 0.51^*$), negatively or with sampling time in centipedegrass ($r^2 = 0.73^{**}$), tall fescue ($r^2 = 0.82^{***}$), and zoysiagrass ($r^2 = 0.56^*$) positively. The increase and/or decrease of C over time (Figure 27) could be related to microbial biomass decomposition and C release in soil and/or immobilization by microbial population.

Nitrogen Release from Turfgrass Clippings

Nitrogen release data were fit to double exponential decay models on a normalized basis. Adjusted R² value for that model was high, with the exception of that calculated for zoysiagrass (0.69). This was likely due to the small amount of N and fast release of labile N for that turfgrass.

Initial N concentrations in turfgrasses were low, with an N content of 14.5 g kg⁻¹ in centipedegrass, compared to 14.7 and 14.6 g kg⁻¹ in St. Augustinegrass and zoysiagrass, respectively. Tall fescue had a higher N concentration (40 g kg⁻¹), followed by bermudagrass (23 g kg⁻¹). After 24 weeks of field incubation, zoysiagrass had lost

17.3% of N content, compared to 47.4, 31.1, 49.5, and 83.6% under bermudagrass, centipedegrass, St. Augustinegrass, and tall fescue, respectively (Figure 28).

The labile decay constant of centipedegrass (0.88; Table 15) was greater than that of bermudagrass, tall fescue, and zoysiagrass combined. The recalcitrant decay constant of centipede (0.01) was not greater than that of tall fescue (0.01) and more than 100 times greater than zoysiagrass (3.62×10^{-19}). The higher the constant the faster the decomposition will be. The labile portion of tall fescue was slightly greater than 72%, and the recalcitrant portion was lower than 23% (Table 15). McCurdy et al., (2013) found similar results, with a labile fraction of 80% and a recalcitrant portion of 25% in white clover litter incubated under field conditions. In contrast, centipedegrass had a low labile fraction (9%) and a high recalcitrant fraction (91%). The higher the labile portion the faster that decomposition occurs.

After 32 weeks, tall fescue N declined 84%, from an initial equivalent of 360 g N m^{-2} to 58 g N m^{-2} , while zoysiagrass N decreased 17.7%, to 296 g N m^{-2} . After 46 weeks there were significant differences in N among turfgrasses. The only non-significant N content was between tall fescue and St. Augustinegrass, and between bermudagrass and zoysiagrass (Table 19).

For the five turfgrass species, the relationship between biomass and N content over time was always significant (Table 20; $P < 0.05$). As shown in this data, a linear relationship between percent remaining biomass and nutrient concentration is common in other plants (Aber and Melillo, 1980; Blair, 1988; Gallardo and Merino, 1992). Changes in mass indicate loss of organic C during respiration, while changes in N concentration

indicate changes in the microbial biomass (Aber and Melillo, 1980). Table 20 and Figure 29 show strong correlation between biomass content and remaining N for all turfgrasses. Remaining biomass at each sampling was significantly correlated with N in all species. Table 21 and Figure 30 illustrate the change in N concentration over time, as affected by turfgrass species. In all cases N concentration in tissue increased over time. Previous studies reported similar results under different ecosystems, included five exotic plant species such as *Acacia auriculiformis*, *Cassia siamea*, *Casuarina equisetifolia*, *Eucalyptus hybrid* and *Gravelia pteridifolia* growing on coal mine spoil (Dutta and Agrawal, 2001), in pine needles (Enoki and Hawaguchi, 2000), woodland ecosystems (Guo and Sims, 1999) and a mixture of herbaceous plants in Mediterranean subtropical agro-ecosystems (Rodriguez Pleguezuelo et al., 2009). This N increase could be related to lignin compounds. Fioretto et al., (2005) reported that most N in litter that has 65 to 95% total N is from lignin compounds, and N release will take place when lignin decomposition happens. They also suggested that lignin structure covers cell wall proteins, which protect them from microbial degradation (Fioretto et al., 2005).

C:N Ratio

The nature of warm season turfgrass decomposition is different than cool season turfgrass (USDA-NRCS, 2004). Warm season turfgrass contains a relatively high concentration of C, allowing for slower initial decay (Fortes et al., 2012). Initial C:N ratios were 29.3 for zoysiagrass, 18.6 for bermudagrass, 28.7 for centipedegrass, and 22.4 for St. Augustinegrass. The C:N ratio of tall fescue was 10.3, in comparison. Table 22 illustrates C:N ratios of the turfgrass clippings. C:N ratios are typically used to describe a residue's propensity to immobilize and/or mineralize soil inorganic N (Hadas

et al., 2004). The critical C:N ratio for immobilization and/or mineralization is between 19 and 30 (Lutz and Chandler, 1946; Aber and Melillo, 1980; Edmonds, 1984). Thus, the C:N ratio plays a major role on N dynamics in soil (Magid et al., 1997; Hadas et al., 2002).

During the first 4 weeks the C:N ratio increased for all turfgrass species, with the exception of centipedegrass. Thereafter the C:N ratio declined. After 46 week the C:N ratio of tall fescue was 9.6, compared to 13.4, 14.3, 15.0, and 20.5 for St. Augustinegrass, bermudagrass, centipedegrass, and zoysiagrass, respectively (Table 22). With C:N ratios of nearly 10 throughout for the decomposition for tall fescue, this turfgrass was likely to be a long-term N contributor. In contrast, with C:N ratios of ≥ 24 throughout for the decomposition (early decomposition of centipedegrass up to 8 weeks, Table 22), this turfgrass was likely responsible for immobilization of N from soil.

Lignin/N Ratio

Warm season turfgrasses had higher lignin:N ratio contents than the cool season turfgrass tall fescue. Initial lignin:N ratios were determined to be 3.4 (bermudagrass), 2.7 (centipedegrass), 12.5 (St. Augustinegrass), 4.1 (zoysiagrass) and 0.8 (tall fescue). Lignin:N ratios of warm season turfgrasses are comparable with that measured for subtropical forest ecosystem litter (Laishram and Yadava, 1988). The low lignin of tall fescue helps to explain the faster decomposition of tall fescue clippings compared to other turfgrasses. Similar conclusions were drawn by Sterjiades and Erikson (1993), Hobbie (1996), Kitayama et al (2004) and Li-Hu et al (2014). The litter exhibiting higher initial lignin content had slower decomposition rates. For example, decomposition of

Quercus dealbata (6% lignin) litter had slower decomposition than *Quercus fenestrata* (4% lignin) (Laishram and Yadava, 1988). Additionally, the concentration of the lignin fraction increased as decomposition proceeded (Berg and McClaugherty, 2000) and the litter becomes enriched with lignin (Devi and Yadava, 2007). In later stages of decomposition, recalcitrant substances become more dominant in the residue, and in some cases the rate of decomposition approaches zero.

Conclusions

Our research demonstrates important aspects of warm and cool season turfgrass decomposition, mainly that tall fescue is comprised mostly of a quickly decaying labile fraction. Labile and recalcitrant decomposable C and N are important for short- and/or long-term effects on available N concentration in soil, as well as soil C sequestration. Modeling warm and cool season turfgrass decomposition may enable turfgrass researchers and professionals to more accurately predict nutrient contribution to underlying soils and subsequent years of production. Decomposability of C and N may have a direct effect on the timing and quantity of N fertilizer application. Such information could be used to better predict N cycling and use in different turfgrasses. Moreover, differences in soil C sequestration under southeastern U.S. turfgrass species can be explained in large part by differences in tissue decomposition among those species.

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Table 15. Double exponential decay equations regressed on time (weeks) for mass, C, and N-loss from turfgrass incubated in litter bags under field conditions. Double exponential decay equations are in the form of $Y = Ae^{-bx} + Ce^{-dx}$, where Y = remaining mass (normalized %), A = the labile portion, C = the recalcitrant portion, b and d are the labile and the recalcitrant constants, respectively, and x = time in weeks.

Turfgrass	Equation	P > F [†]	R²_{adj}	Syx [‡]
Mass				
Bermudagrass	$Y = 56.22e^{-0.08x} + 44.33e^{-0.003x}$	<0.0001	0.99	1.14
Centipedegrass	$Y = 89.54e^{-0.05x} + 13.75e^{-8.28E-19x}$	<0.0001	0.97	5.24
St. Augustinegrass	$Y = 19.43e^{-0.09x} + 80.02e^{-0.02x}$	<0.0001	0.96	4.69
Tall Fescue	$Y = 91.25e^{-0.11x} + 9.47e^{-8.12E-19x}$	<0.0001	0.96	4.69
Zoysiagrass	$Y = 16.87e^{-0.12x} + 84.14e^{-0.01x}$	<0.0001	0.96	4.69
Carbon				
Bermudagrass	$Y = 47.61e^{-0.11x} + 52.13e^{-0.01x}$	<0.0001	0.99	2.46
Centipedegrass	$Y = 87.82e^{-0.04x} + 13.72e^{-4.93E-18x}$	<0.0001	0.98	3.14
St. Augustinegrass	$Y = 87.67e^{-0.04x} + 5.21e^{-9.92E-18x}$	<0.0001	0.96	4.69
Tall Fescue	$Y = 89.91e^{-0.11x} + 9.32e^{-1.08E-17x}$	<0.0001	0.97	6.04
Zoysiagrass	$Y = 11.17e^{-0.28x} + 90.14e^{-0.01x}$	<0.0001	0.98	2.06
Nitrogen				
Bermudagrass	$Y = 30.15e^{-0.28x} + 69.82e^{-0.01x}$	<0.0001	0.98	2.97
Centipedegrass	$Y = 8.64e^{-0.89x} + 91.36e^{-0.01x}$	<0.0001	0.97	2.54
St. Augustinegrass	$Y = 28.58e^{-0.56x} + 72.31e^{-0.02x}$	0.0002	0.95	4.70
Tall Fescue	$Y = 72.26e^{-0.21x} + 22.68e^{-0.01x}$	0.0001	0.96	5.71
Zoysiagrass	$Y = 20.49e^{-0.16x} + 79.94e^{-3.62E-19x}$	0.0290	0.69	4.95

[†] Significance of fit.

[‡] Standard error of the estimate.

Table 16. Comparison of dry mass weight percent at the end of the study (46 weeks).

Comparison	Difference Between Means	95% Confidence Limits		
Tall Fescue - St. Augustinegrass	14.6	8.2	20.9	***
Tall Fescue - Centipedegrass	20.2	13.9	26.6	***
Tall Fescue - Bermudagrass	32.1	25.8	38.5	***
Tall Fescue - Zoysiagrass	39.0	32.6	45.3	***
St. Augustinegrass - Tall Fescue	-14.6	-20.9	-8.2	***
St. Augustinegrass - Centipedegrass	5.6	0.4	10.8	***
St. Augustinegrass - Bermudagrass	17.5	12.3	22.7	***
St. Augustinegrass - Zoysiagrass	24.4	19.2	29.5	***
Centipedegrass - Tall Fescue	-20.2	-26.6	-13.9	***
Centipedegrass - Augustinegrass	-5.6	-10.8	-0.4	***
Centipedegrass - Bermudagrass	11.9	6.7	17.0	***
Centipedegrass - Zoysiagrass	18.7	13.5	23.9	***
Bermudagrass - Tall Fescue	-32.1	-38.5	-25.8	***
Bermudagrass - St. Augustinegrass	-17.5	-22.7	-12.3	***
Bermudagrass - Centipedegrass	-11.9	-17.1	-6.7	***
Bermudagrass - Zoysiagrass	6.8	1.6	12.1	***
Zoysiagrass - Tall Fescue	-39.0	-45.3	-32.6	***
Zoysiagrass - St. Augustinegrass	-24.4	-29.5	-19.2	***
Zoysiagrass - Centipedegrass	-18.7	-23.9	-13.5	***
Zoysiagrass - Bermudagrass	-6.8	-12.1	-1.6	***

Comparisons significant at the 0.05 level are indicated by ***.

Table 17. Comparison of remaining C content at the end of the study (46 weeks).

Comparison	Difference Between Means	95% Confidence Limits		
Tall Fescue - St. Augustinegrass	6.5	0.5	12.5	***
Tall Fescue - Centipedegrass	21.7	15.7	27.8	***
Tall Fescue - Bermudagrass	24.6	18.6	30.6	***
Tall Fescue - Zoysiagrass	42.8	36.8	48.9	***
St. Augustinegrass - Tall Fescue	-6.5	-12.5	-0.5	***
St. Augustinegrass - Centipedegrass	15.2	10.3	20.1	***
St. Augustinegrass - Bermudagrass	18.1	13.2	23.0	***
St. Augustinegrass - Zoysiagrass	36.3	31.4	41.2	***
Centipedegrass - Tall Fescue	-21.7	-27.8	-15.7	***
Centipedegrass - Augustinegrass	-15.2	-20.1	-10.3	***
Centipedegrass - Bermudagrass	2.8	-2.1	7.7	
Centipedegrass - Zoysiagrass	21.1	16.1	26.0	***
Bermudagrass - Tall Fescue	-24.6	-30.6	-18.6	***
Bermudagrass - St. Augustinegrass	-18.1	-23.0	-13.2	***
Bermudagrass - Centipedegrass	-2.8	-7.7	2.1	
Bermudagrass - Zoysiagrass	18.2	13.3	23.1	***
Zoysiagrass - Tall Fescue	-42.8	-48.9	-36.8	***
Zoysiagrass - St. Augustinegrass	-36.3	-41.2	-31.4	***
Zoysiagrass - Centipedegrass	-21.1	-26.0	-16.1	***
Zoysiagrass - Bermudagrass	-18.2	-23.1	-13.3	***

Comparisons significant at the 0.05 level are indicated by ***.

Table 18. Linear regression equations regressed on C concentration (%) from turfgrass incubated in litter bags under field conditions.

Turfgrass	Equation	F	P	R²	Syx[‡]
Bermudagrass	$Y = 39.4 - 0.08x$	24.8	0.0016	0.78	0.76
Centipedegrass	$Y = 41.5 + 0.09x$	18.9	0.0033	0.73	0.97
St. Augustinegrass	$Y = 34.1 - 0.10x$	7.2	0.0312	0.51	1.7
Tall Fescue	$Y = 40.4 + 0.11x$	32.5	0.0007	0.82	0.9
Zoysiagrass	$Y = 42.6 + 0.08x$	8.9	0.0201	0.56	1.2

[‡] Standard error of the estimate.

Table 19. Comparison of remaining N content at the end of the study (46 weeks).

Comparison	Difference Between Means	95% Confidence Limits		
Tall Fescue - St. Augustinegrass	13.3	-0.9	27.5	
Tall Fescue - Centipedegrass	32.2	17.9	46.4	***
Tall Fescue - Bermudagrass	48.4	34.2	62.7	***
Tall Fescue - Zoysiagrass	54.3	40.1	68.5	***
St. Augustinegrass - Tall Fescue	-13.3	-27.5	0.9	
St. Augustinegrass - Centipedegrass	18.8	7.2	30.5	***
St. Augustinegrass - Bermudagrass	35.1	23.5	46.7	***
St. Augustinegrass - Zoysiagrass	40.9	29.3	52.6	***
Centipedegrass - Tall Fescue	-32.2	-46.4	-17.9	***
Centipedegrass - Augustinegrass	-18.8	-30.5	-7.2	***
Centipedegrass - Bermudagrass	16.2	4.6	27.8	***
Centipedegrass - Zoysiagrass	22.1	10.4	33.7	***
Bermudagrass - Tall Fescue	-48.4	-62.7	-34.2	***
Bermudagrass - St. Augustinegrass	-35.1	-46.7	-23.5	***
Bermudagrass - Centipedegrass	-16.2	-27.8	-4.6	***
Bermudagrass - Zoysiagrass	5.8	-5.7	17.4	
Zoysiagrass - Tall Fescue	-54.3	-68.5	-40.1	***
Zoysiagrass - St. Augustinegrass	-40.9	-52.6	-29.3	***
Zoysiagrass - Centipedegrass	-22.1	-33.7	-10.4	***
Zoysiagrass - Bermudagrass	-5.8	-17.4	5.7	

Comparisons significant at the 0.05 level are indicated by ***.

Table 20. Linear regression equations regressed on N and remaining mass from turfgrass incubated in litter bags under field conditions.

Turfgrass	Equation	P > F [†]	R²_{adj}	Syx [‡]
Bermudagrass	Y = -2.0 + 1.1x	<0.0001	0.95	5.31
Centipedegrass	Y = 45.1 + 0.5x	<0.0001	0.91	4.59
St. Augustinegrass	Y = 9.0 + 0.8x	<0.0001	0.90	7.17
Tall Fescue	Y = 2.4 + 0.8x	<0.0001	0.94	7.61
Zoysiagrass	Y = 53.3 + 0.4x	0.0064	0.63	5.34

[†] Significance of fit.

[‡] Standard error of the estimate.

Table 21. Linear regression equations regressed on N concentration (%) from turfgrass incubated in litter bags under field conditions.

Turfgrass	Equation	F	P	R²	Syx[‡]
Bermudagrass	$Y = 2.0 + 0.02x$	20.4	0.0028	0.74	0.16
Centipedegrass	$Y = 1.4 + 0.04x$	143.8	<0.0001	0.95	0.16
St. Augustinegrass	$Y = 1.4 + 0.02x$	50.5	0.0002	0.88	0.15
Tall Fescue	$Y = 3.4 + 0.04x$	10.1	0.0157	0.59	0.55
Zoysiagrass	$Y = 1.4 + 0.02x$	89.9	<0.0001	0.93	0.09

[†] Significance of fit.

[‡] Standard error of the estimate.

Table 22. Carbon:nitrogen ratios of turfgrasses at each sampling period.

Week	Bermuda grass	Centipede grass	Tall fescue	St. Augustine grass	Zoysia grass
0	18.6±0.7	28.7±1.0	10.3±0.1	22.4±1.3	29.3±0.7
1	19.4±0.3	29.1±0.8	11.6±0.3	22.2±2.5	28.3±0.5
2	19.9±0.4	30.5±1.3	12.4±0.2	23.2±0.9	30.5±0.6
4	20.6±0.5	28.7±1.4	14.0±0.4	24.7±1.0	28.6±1.1
8	19.4±0.3	26.5±0.5	12.8±0.5	18.3±5.5	28.3±0.6
16	16.1±0.9	22.1±0.5	8.9±0.1	21.7±1.7	26.8±0.5
24	15.1±0.4	17.8±0.7	9.4±0.4	16.4±0.8	26.1±1.1
32	14.7±0.2	15.5±0.4	8.8±0.4	14.7±0.3	23.6±0.4
46	14.3±0.5	15.0±0.2	9.6±0.6	13.4±0.3	20.5±0.3

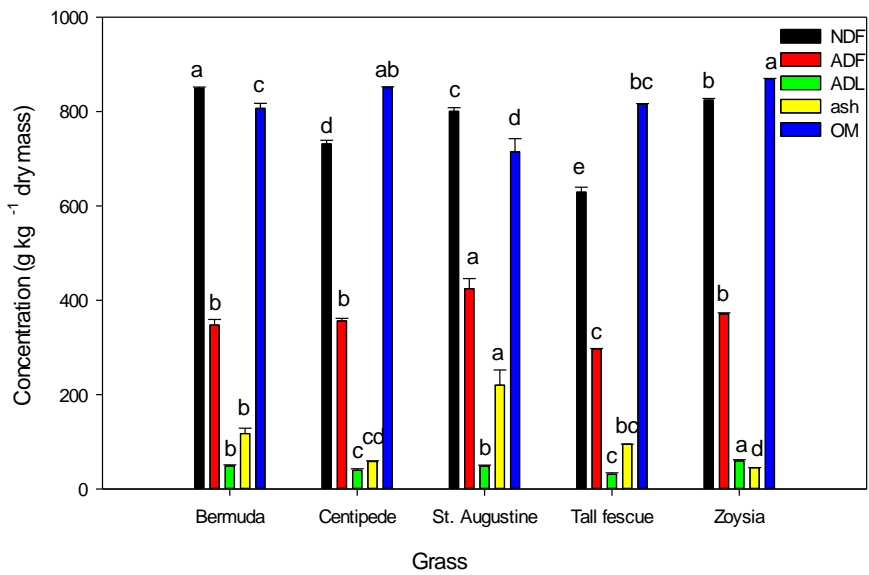


Figure 22. Initial (at harvest) fiber analysis of turfgrasses grown and decomposed at the Auburn University Turfgrass Research Unit dry matter basis. Where, NDF = Natural Detergent Fiber, ADF = Acid Detergent Fiber, and ADL = Acid Detergent Lignin. Bars with similar letters within each analyte indicate no significant difference between those means. Error bars represent standard errors about the mean.

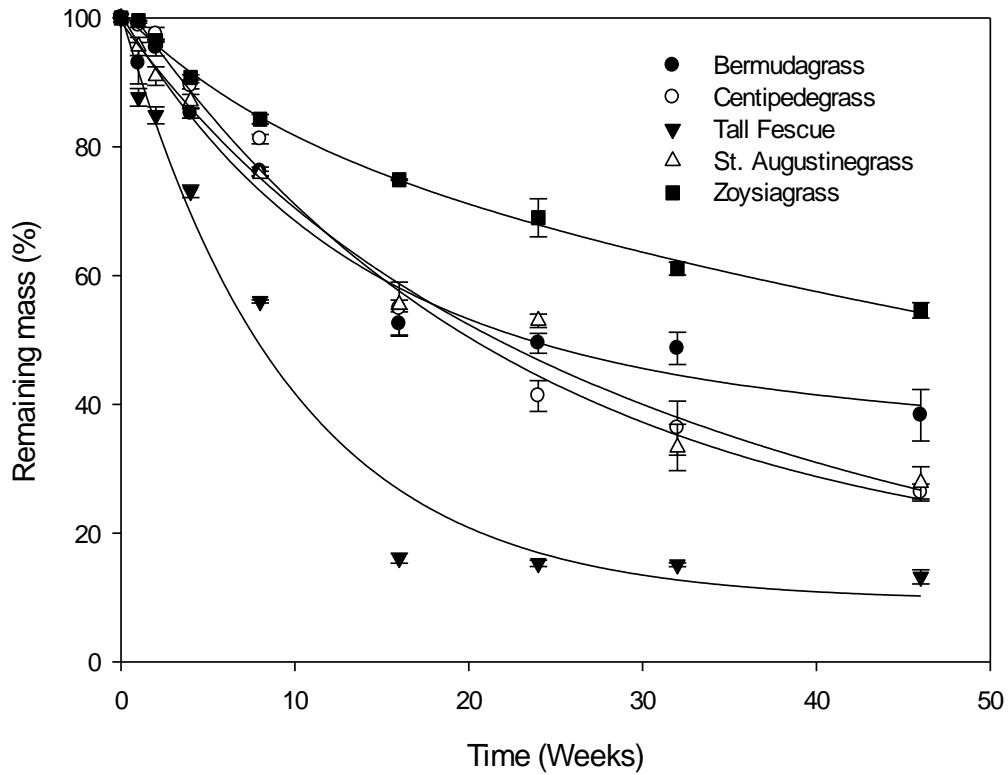


Figure 23. Mass loss from turfgrass clippings over time, Auburn University Turfgrass Research Unit, 2012-2013. Lines represent fitted curves for each grass describing decay pattern over time. Bars represent standard errors about the mean.

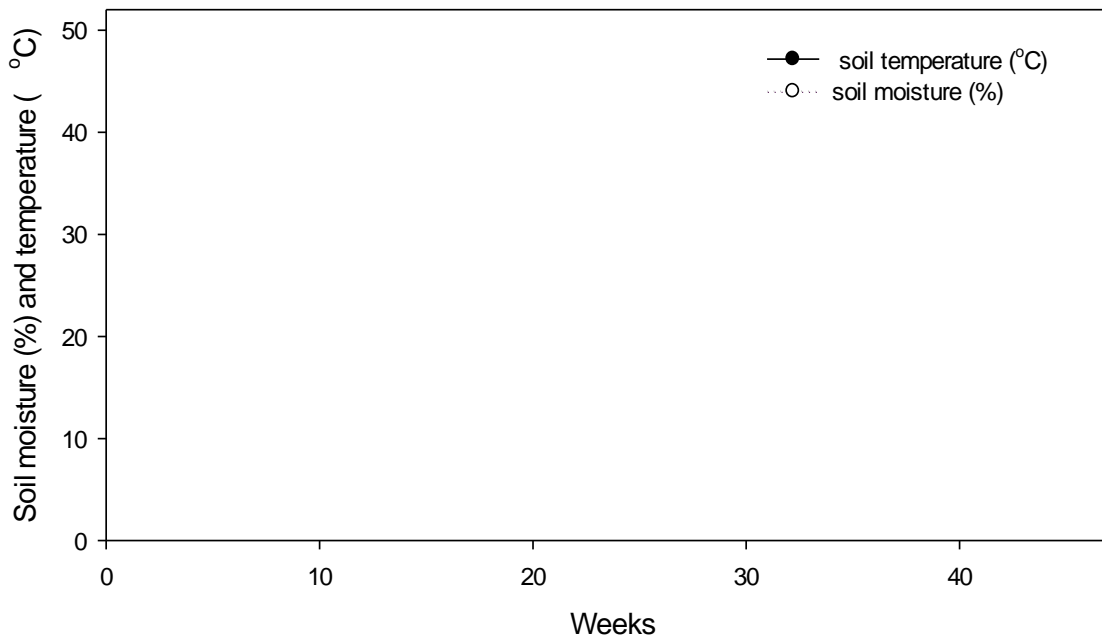


Figure 24. Soil moisture and temperature at 10 cm depth, Auburn, AL, 2012-2013.

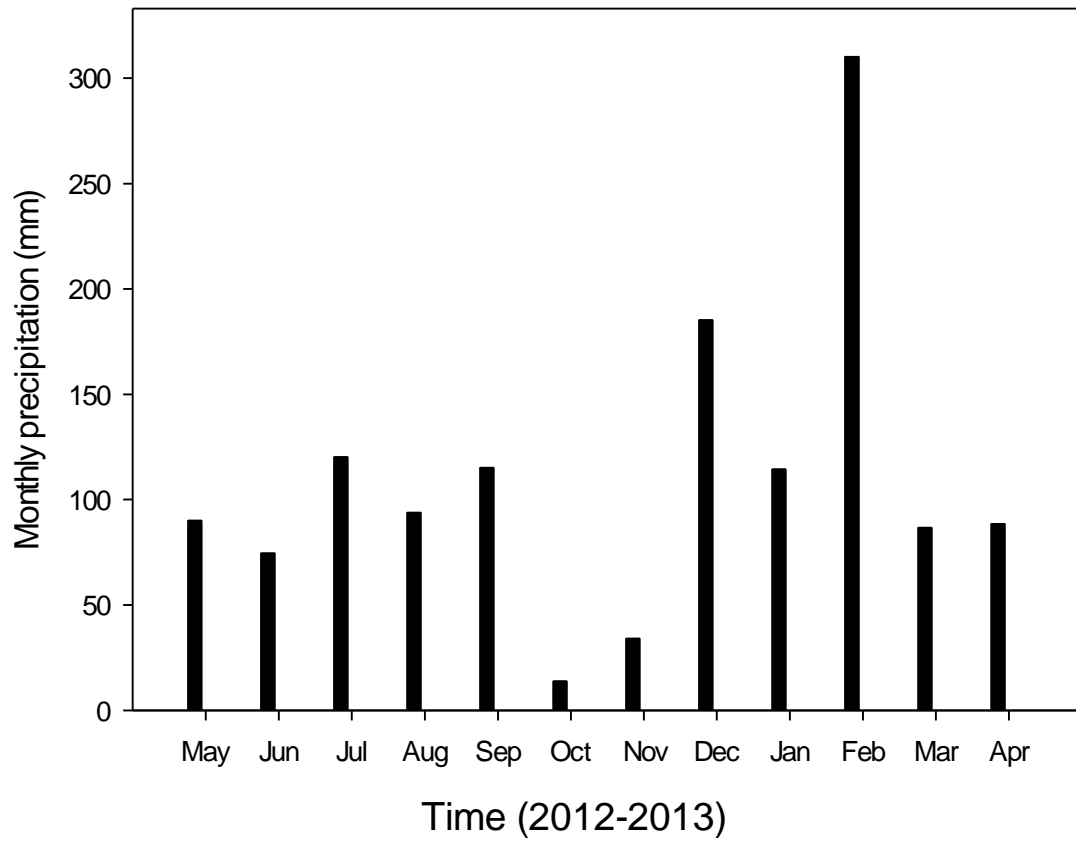


Figure 25. Monthly precipitation at the Auburn turfgrass experimental station

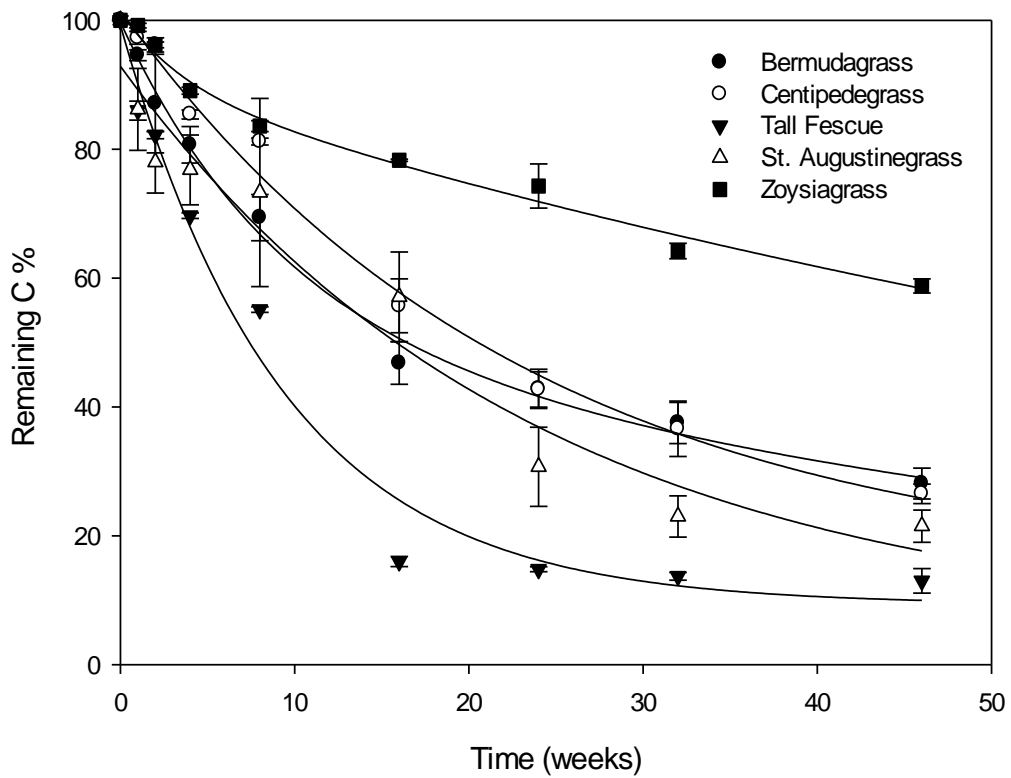


Figure 26. Carbon loss from turfgrass clippings, Auburn university Turfgrass Research Unit, 2012-2013. Lines represent fitted curves for each grass describing decay pattern over time. Bars represent standard errors about the mean.

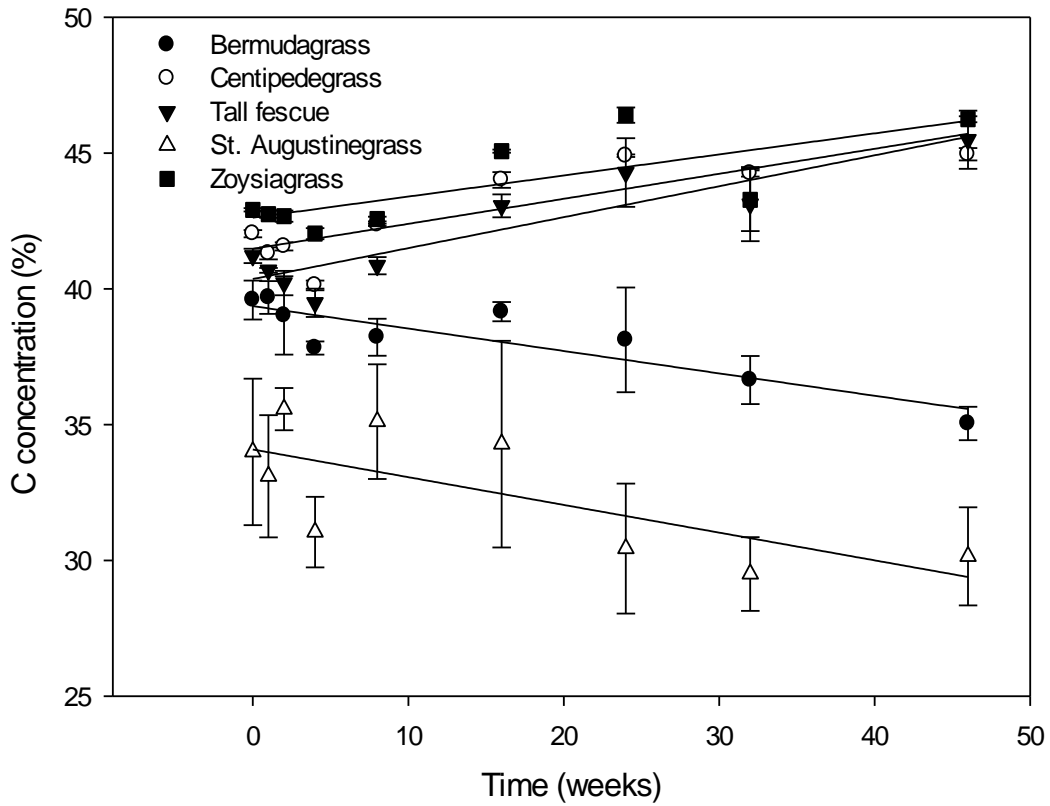


Figure 27. Carbon concentration (%) in harvested clippings as affected by turfgrass species and time, Auburn University Turfgrass research Unit, 2102-2013. Lines represent fitted linear correlation between C% and time of sampling for each grass. Bars represent standard errors about the mean.

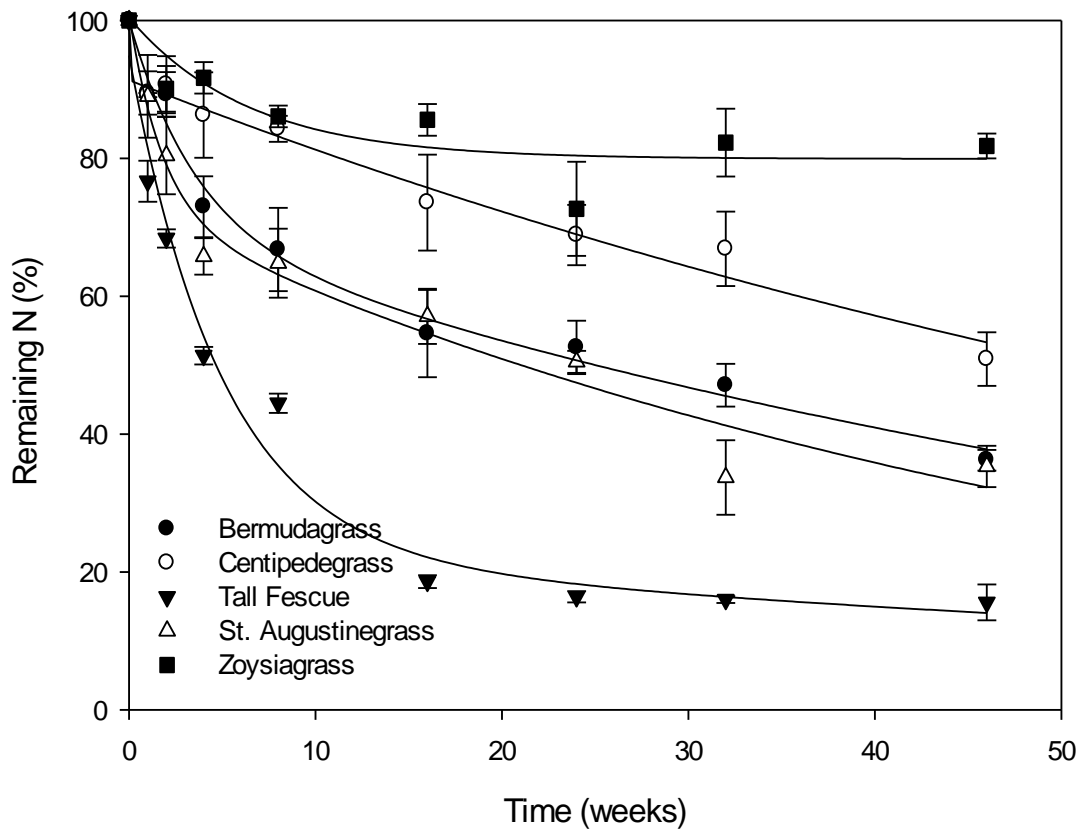


Figure 28. Nitrogen loss from turfgrass clippings as affected by turfgrass species and sampling time, Auburn University Turfgrass Research Unit, 2012-2013. Lines represent fitted curves for each grass describing decay pattern over time. Bars represent standard errors about the mean.

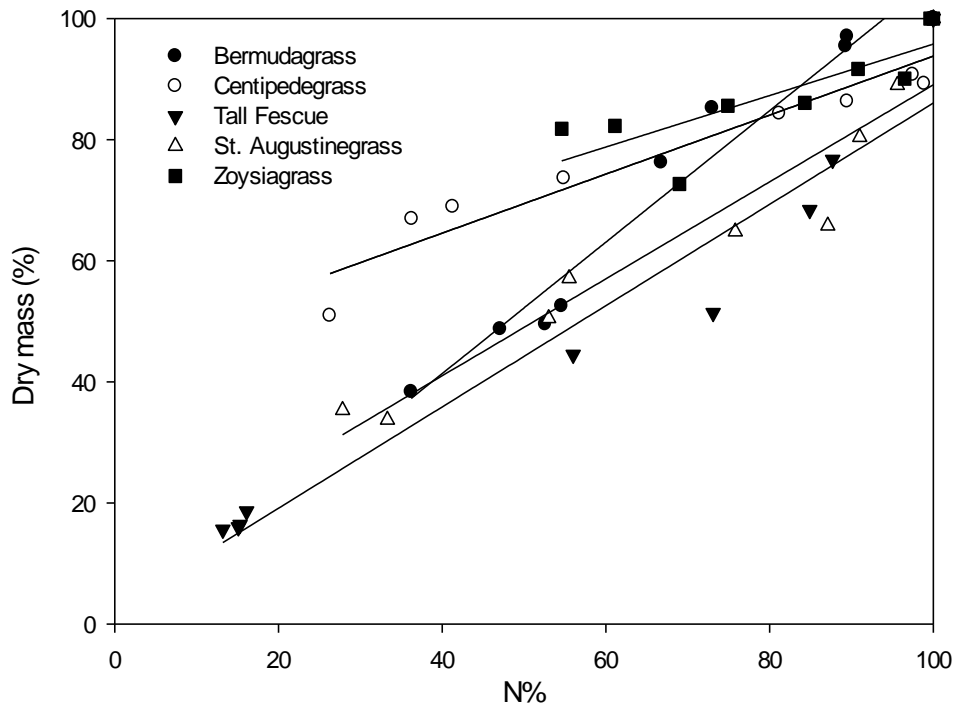


Figure 29. Linear relationship between N content and dry mass as affected by turfgrass species, Auburn University Turfgrass Research Unit, 2012-2013. Lines represent linear correlation of dry mass of each grass with N%. Bars represent standard errors about the mean.

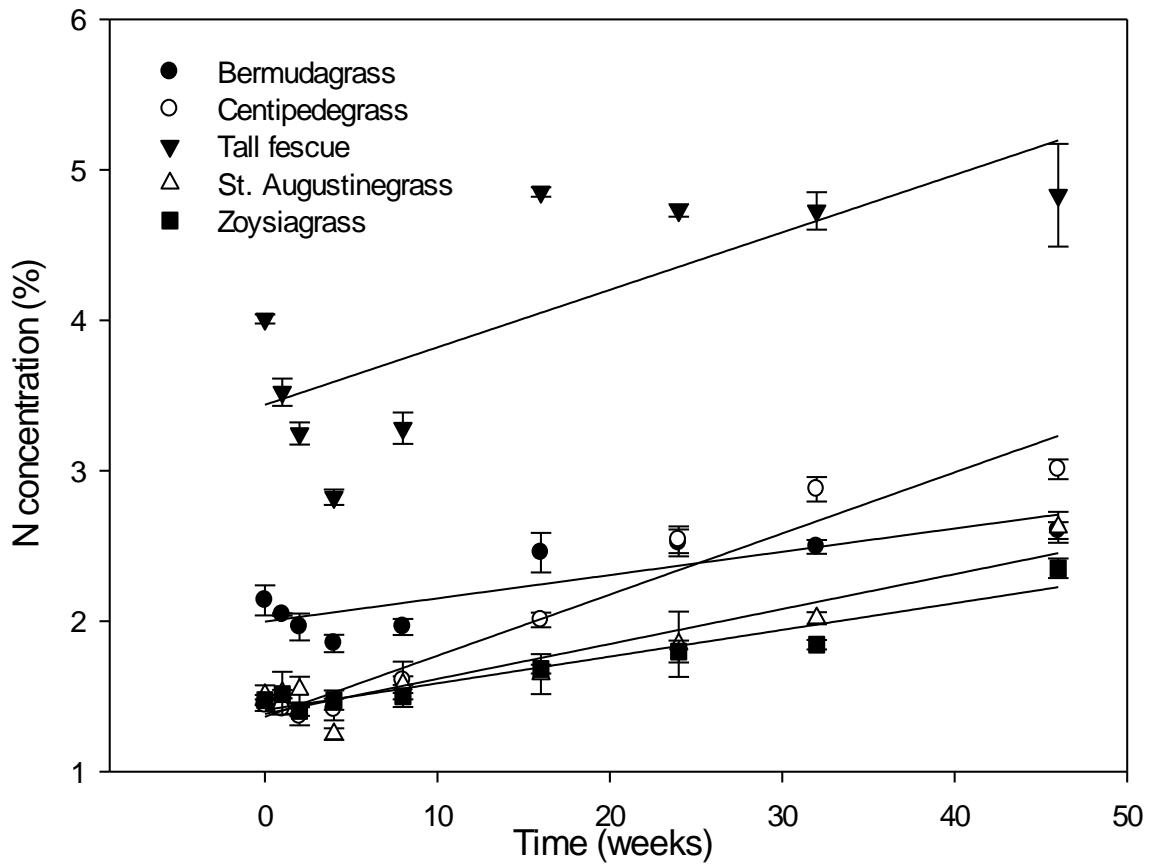


Figure 30. Nitrogen concentration (%) in harvest clippings as affected by time and turfgrass species, Auburn University Turfgrass Research Unit, 2012-2013. Lines represent linear correlation between N% and time of sampling of each grass. Bars represent standard errors about the mean.